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Version: Version of Record

Link(s) to article on publisher's website:

<http://dx.doi.org/doi:10.21954/ou.ro.0000e290>

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COMPOSTING AND VERMICOMPOSTING WASTE PAPER SLUDGE

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Thesis submitted for the degree of Doctor of Philosophy

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30 September 1999

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DATE OF SUBMISSION: 30 SEP 1999

DATE OF AWARD: 22 APRIL 2000

Abstract

Increasing legislative and economic pressure to find more sustainable methods of organic waste management has fuelled innovation in biological treatment technology. By-products of paper manufacturing industries provide a large source of organic waste, which is known to have a high environmental impact. This waste paper sludge has been shown to be amenable to biological treatment. Recent research has confirmed that windrow-composting and vermicomposting techniques have potential to treat these wastes and share many economic and environmental benefits. Many authors have suggested that sludge specific composting methods need to be developed and this research aims to provide fundamental data in this respect. The treatment of specific waste paper sludges was investigated through small and large scale experiments with the aim of optimising these processes with minimal intervention.

Identical samples of a selected waste paper sludge feedstock were used in large scale investigations into the application of each composting technique, and the performance of each process and resulting products was evaluated.

Windrow composting and vermicomposting were found to stabilise and enhance waste paper sludge in very different ways, producing unique products. In terms of processing, windrow composting resulted in more rapid rates of stabilisation and although the performance of the vermicomposting process was less effective in these respects, it afforded additional benefits as a treatment of waste paper sludge.

Both processes were found to stabilise and enhance waste paper sludge but the selection of one system or the other will depend largely on the objectives of the project and the criteria required of the finished product.

Acknowledgements

Firstly I would like to thank the ESRC and Original Organics for funding my research, and the Open University for providing the facilities with which to conduct it.

Special thanks goes to Jim Frederickson and Dick Morris for their kind assistance and patient guidance.

Finally, thank you to all those with whom I shared a room for putting up with nasty smells and making life so much easier.

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1. Introduction

1.1 Introduction

The importance of microbial activity in recycling organic matter within the biosphere is widely known; and the exploitation of this natural phenomenon to dispose of unwanted organic matter, e.g., sewage, has long been established (Postgate 1986). Investigations into the further utilisation of microbial processes to treat a variety of biodegradable wastes are well documented. These fall into two main categories: anaerobic processes, e.g., methane production, or anaerobic digestion (Hobson *et al.* 1981); and aerobic processes, e.g., composting (Diaz *et al.* 1993 and De Bertoldi *et al.* 1996). This study provides a comparative investigation of two aerobic processes used to treat waste paper sludge: windrow-composting, a traditional composting method (Golueke & Diaz 1996); and vermicomposting, composting with the assistance of earthworms (Neuhauser *et al.* 1979). Although considerable research has been conducted on both these separate aerobic processes, the literature comparing these two processing options for the treatment of waste paper sludge is sparse.

This introductory Chapter sets out the context of organic waste management in the UK and identifies the factors that helped to determine the aims of this study. The main aims of the research are then detailed, followed by a thesis outline.

Biological waste treatment methods are receiving increasing recognition within EU and UK waste policy. Waste disposal options in the UK are increasingly subject to legislative measures, which provide incentives for the investigation and adoption of alternative waste treatment options including composting. This policy and legislative context is outlined in section 1.2.

Section 1.3 provides a brief review of the benefits of windrow composting and vermicomposting over organic waste disposal, a more extensive literature review of each process is given in the relevant Chapters.

Section 1.4 highlights the appropriateness of the UK paper manufacturing industry for the application of composting techniques in treating organic waste. Biological waste treatment as part of an integrated waste management system is briefly considered in section 1.5.

The main aims of this research are given in section 1.6, which is followed by a thesis outline in section 1.7.

1.2 Overview of UK Waste policy and legislation

1.2.1 Introduction

The world has witnessed a considerable increase in environmental concerns over the last 30 years (Hunt & Johnson 1995). An 'Environmental Revolution' (Tyler-Miller 1994) fuelled by a complex array of influences, such as, the public, the media, and pressure groups (Hunt & Johnson 1995). A surge in environmental thinking led to the evolution of a number of key environmental concepts, e.g., 'Sustainable Development', defined as, 'development that meets the needs of the present without compromising the ability of future generations to meet their own needs' (WCED 1987); and the OECD's 'Polluter Pays Principle', which states, 'the polluter should bear the expenses of carrying out [pollution prevention and control] measures decided by public authorities to ensure the environment is in an acceptable state' (Markandya & Richardson 1992). The environment is increasingly considered a finite resource whose use or degradation should incur a charge (e.g., Pearce *et al.* 1989).

Solid waste management is an issue of major concern in achieving environmentally sound and sustainable development (section 2, Chapter 21, Agenda 21). Waste management has important implications for both reducing pollution, and conserving resources (White *et al.* 1995). Waste legislation within the UK and EU seemingly reflects new environmental concerns and concepts.

1.2.2 Waste: a pollutant and an indicator of resource conservation

Environmental legislation in the UK and EU has historically focused on regulation and control of environmental pollution from harmful substances (Hunt & Johnson 1995). Unsurprisingly in the UK solid waste management was regulated in terms of its polluting aspects, e.g., the Control of Pollution Act 1974. Again due to the threat of pollution, much attention has been paid to hazardous and toxic wastes (e.g., EC 1978 & 1991c), known as 'special waste' in the UK (DoE 1980 & 1996). However, waste is considered increasingly as a resource and an indicator of resource usage. 'Waste' is a proposed 'headline indicator' in the UK Governments new Sustainable Development Strategy; 'waste' used as a, 'measure of the efficient use of resources' (DETR 1998a).

1.2.3 Legislative approaches to waste management

Most waste legislation in the UK and EU focuses upon 'end-of-pipe' regulation and control of existing waste management systems, e.g., EC Directives on the prevention of air pollution by waste incineration plants (EC 1989a,b) and UK controlled waste regulations (DoE 1992). These regulatory standards are beginning to be complemented by strategic policy and

legislative measures, such as, waste recovery targets and financial incentives/deterrents; 'market based instruments' designed to change the ways in which waste is managed in the future (White *et al.* 1995).

1.2.4 Waste legislation in the nineties

The late eighties saw a phenomenal growth in environmental concern (Hunt & Johnson 1995), which may explain a surge of waste management legislation seen in the UK and EU during the 1990s. The early to mid. 1990s witnessed more stringent definitions of wastes, more tightly regulated waste management practices, increased licensing, and an increased emphasis on waste reduction and recovery.

The publication of the EU policy document, 'Waste Management Strategy' (EC 1989c), advocated the prevention of waste production as well as waste reuse and recovery; the polluter pays principle; and the proximity principle (waste should be disposed of as close to its source of production as possible). This led to the revision of the 1975 EC Waste Directive in 1991 (EC 1991a). This was soon complemented by the EC Directive on the Landfilling of waste (EC 1997) and the Packaging Waste Directive (EC 1994) to discourage waste production and disposal.

In 1990 the UK saw the publication of the white-paper, 'This Common Inheritance' (DoE 1990a), which explicitly outlined the Government's thinking on environmental issues, including waste management. It also proposed the UK's first waste recycling target - 25% of household waste by the year 2000.

The white-paper was later placed in a legislative framework, the UK's Environmental Protection Act (EPA) (DoE 1990b). Though the 1990 EPA did not specify any waste recycling targets, it did call for the drafting of recycling plans by waste collection authorities (WCAs); WCAs could receive payment in the form of 'recycling credits' for waste diverted for recycling (including composting). The 1990 EPA re-enacted the provisions of the Control of Pollution Act 1974 (relating to waste disposal on land), adding tighter regulations concerned with the collection and disposal of waste. This included the creation of a waste management licensing system, and the introduction of the 'Duty of Care' law (DoE 1991). The 'Duty of Care' outlined obligations of any persons dealing with 'controlled waste' (household, commercial or industrial waste) to ensure its safe disposal (DoE 1991).

UK white-paper Making Waste Work 1995

This new wave of waste legislation contributed to the UK's first white-paper specifically targeting waste management, 'Making Waste Work' (DoE 1995). This was based upon ideas outlined in the UK Government's Sustainable Development Strategy (DoE 1994). Making

Waste Work outlined the UK's 'waste hierarchy', a list of waste treatment options ranked according to their sustainability. Waste reduction was placed as the most desirable option, followed by material or energy recovery, and finally disposal; resembling the hierarchy supported within the EU (EC 1989c). In line with the waste hierarchy, the white-paper set out a number of targets for reductions in waste disposal and increased waste recovery. In addition to including composting as a means of achieving these targets, Making Waste Work set a target to compost one million tonnes of organic waste by the year 2000. Following expansion in the composting industry over the last few years it is almost certain that this composting target will be met (Composting Association, 1999), highlighting the increasing recognition of composting as a valid management option for waste organics.

BPEO (Best Practicable Environmental Option)

Though the UK Government supports the principle of the waste hierarchy, it is considered as a guide, 'not a prescriptive set of rules' (DETR 1998b). The UK Government advocates the BPEO (Best Practicable Environmental Option) approach to choosing waste management options. The Royal Commission on Environmental Pollution defined BPEO as:

'...the outcome of a systematic consultative decision making procedure which emphasises the protection and conservation of the environment across land, air and water. The BPEO procedure establishes, for a given set of objectives, the option that provides the most benefit or least damage to the environment as a whole, at acceptable cost, in the long term as well as short term' (RCEP 1988).

For example, factors such as the 'proximity principle' (disposing of waste as near to the source of production as possible) must be considered when choosing waste management systems; options lower in the waste hierarchy may have much lower transport requirements than more 'desirable' options, significantly reducing economic and, possibly, reducing environmental costs.

Ascertaining and implementing the BPEO for the management of different wastes is a complex task requiring analysis of a large number of technical, economic and environmental factors, leading to the utilisation of a mixture of waste treatment techniques (see section 1.5). Which waste practices are most likely to be the BPEO will also be influenced by national and international legislation.

Market based incentives

Initially, the UK Government employed voluntary promotional strategies to disseminate information on 'best' waste practice. UK industry received particular attention via the Environmental Technology Best Practice Programme (ETBPP), set up in 1994. The ETBPP

was devised to stimulate savings for industry through the encouragement of environmental practices that reduce costs (DoE 1995).

Making Waste Work set out the Government's intention to incorporate market-based instruments into waste management policy to promote the waste hierarchy and help achieve waste targets. Subsequently the Landfill Tax was introduced in 1996, followed by the Packaging Waste Obligations Regulations (DoE 1997). As a consequence, waste producers should now be made to face more of the environmental costs of waste treatment - enforcing the concept of 'producer responsibility', advocated by the UK and EU (e.g., DoE 1995 and EC 1991a respectively).

UK Landfill Tax

The UK Landfill Tax introduced in October 1996, the UK's first primarily environmental tax, imposed an extra cost on waste going to landfill. A levy of £2 per tonne was placed upon 'inert' waste, and £7 per tonne upon 'active' [biodegradable] waste (HMCE 1998). A higher tax level for biodegradable waste reflects environmental costs ('externalities') thought to be associated with the disposal of organic waste at landfill (Pearce & Turner 1993). This tax was increased from £7 to £10 per tonne in April 1999 and is due to increase by £1 per tonne per year over the next 5 years.

With the introduction of the UK landfill tax the Government also set up a landfill tax credit scheme; whereby up to 20% of the tax could be channelled towards bodies with environmental objectives, including research into waste recycling and composting. There have been approximately 50 Environmental Body research projects related to composting funded in this way (Anon. 1998).

EC Directive on Landfill

The EC Directive on Landfill was first proposed in 1991 and the European Council of Ministers reached its first 'common position' in October 1995. This was rejected by the European Parliament in 1996 on the grounds that the Directive provided insufficient environmental protection. Numerous negotiations and amendments led to the adoption of a second common position in June 1998, which was finally adopted in April 1999. The UK now has two years in which to put in place strategies to meet the Directive's targets (Hawkins 1998).

The Directive has profound implications for the way organic waste is managed in the UK. One of the main requirements is the staggered reduction of biodegradable municipal solid waste going to landfill by 25%, 50%, and 65% (based on 1995 figures) within 5, 8, and 15 years after implementation of the Directive into national law (Coggins 1999). Estimates of the

quantities of organic waste that will need to be diverted from landfill vary, but the latest Government draft waste strategy 'A Way With Waste' suggests that this may be as much as 33 million tonnes per year by 2020 (DETR 1999a). Composting has the potential to provide a viable alternative to landfill for some of this diverted waste stream. The Directive also requires the pre-treatment of all municipal waste prior to landfill. Whilst there remains some ambiguity around the finer points of the definition of pre-treatment, it will include biological treatment and composting.

In response to possible impacts of the Landfill Directive, the UK Department of the Environment Transport and the Regions (DETR) set up a Composting Development Group in 1997 to help increase the quantity and quality of municipal waste derived compost produced in the UK (Anon. 1998).

Although the Directive is aimed at municipal biodegradable waste it is likely to have wider implications for all sources of organic waste and emphasises the timely importance of this study.

1.2.5 Future [organic] waste Policy and legislation in the UK and EU

Increasing Landfill Costs

The landfilling of waste currently remains the most viable economic option for the majority of waste produced in the UK. The cost of landfill is set to rise with future increases in the landfill tax (HMCE 1998), and the implementation of the Landfill Directive (Hawkins 1998). Rising landfill costs provide an incentive for developing alternatives to landfill for the treatment of different wastes, including industrial organics.

A study of organic waste land-spreading

Another traditional organic waste disposal route, land-spreading, may also face future restrictions for certain wastes. A ban on sewage disposal at sea in 1998, following the Urban Waste Water Treatment Directive (EC 1991b), coincides with increasing concerns over sewage land-spreading (Coggins 1999). A study of all organic waste land-spreading has recently been commissioned, reflecting concerns of public, animal and plant health; environmental pollution; and soil protection (The Scottish Office 1998).

New UK white-paper on waste

A change in UK Government during 1997 spawned the revision of the UK's waste management strategy, with the publication of a new waste management consultation document: 'Less Waste. More Value' (DETR 1998a). Following this initial consultation exercise the draft strategy 'A Way With Waste' was published in June 1999 (DETR 1999a).

This will lead to a new white-paper on waste management expected in 2000, superseding the previous Government's, 'Making Waste Work' (DoE 1995). A Way With Waste proposes that materials recycling and composting should be considered before energy recovery.

Proposed EC Directive on Composting

The European Commission's Environment Directorate (DGXI) are currently preparing an EC Directive on Composting (Anon. 1998). This will undoubtedly help standardise industry practice, making composting a more viable option for the diversion of organics from landfill. Increased standardisation of composting practice will lead to more consistent and commercially acceptable products. The European Federation of Waste Management has produced a first draft position paper on composting, suggesting the Directive should apply to all organic waste (regardless of source) focusing on environmental protection, and stimulating the use of compost as a soil improver (*op. cit.*). The UK's Environmental Services Association (ESA) has set up a working group to discuss how the Directive could benefit the UK composting industry (*op. cit.*).

1.3 windrow-composting and vermicomposting of organic waste

1.3.1 Introduction

Important research into windrow composting (a traditional form of composting) and vermicomposting are briefly outlined, as more detailed coverage of the literature will be conducted in subsequent Chapters.

Windrow-composting and vermicomposting affect organic matter in a number of ways during processing. These changes afford varying benefits dependant upon the objective of each composting process. The effect of the composting process on organic waste is again briefly outlined, with reference to the economic and environmental value of composting and vermicomposting.

Similarities between the objectives of each process will be highlighted, as will chief differences. A lack of comparable data regarding the processing of similar waste under optimal conditions will also be argued, justifying the need for accurate comparisons of these competing systems.

1.3.2 The windrow composting process

Advances in the understanding of composting process can be dated back to the 1950s, although the utilisation of traditional composting methods to treat organic wastes did not gain stature until the 1970s (Golueke & Diaz 1996). During this period many parameters essential to the composting process were recognised, e.g.:

1. Moisture control – optimum range 55-60% (Goluecke & McGauhey 1953)
2. Temperature control – optimum range 40-60°C (Wylie 1957), maximum temperatures should not exceed 60-65°C (Carpenter 1977)
3. Aeration – oxygen supply and demand (Schulze 1960)
4. Carbon:nitrogen (C:N) and other nutrient ratios – Gray *et al.* 1971

Shortly after this period numerous studies into factors affecting the composting process were conducted, as well as the engineering processes involved (Anon. 1982).

Many types of composting technologies have been developed, based mainly upon the necessity to control temperature and aeration during decomposition (Diaz *et al.* 1993).

Some of the more common composting methods include: use of static piles with forced aeration, mechanically-turned windrows, and in-vessel composting systems (op. cit. 1993).

This study employs the more traditional mechanically-turned windrow (pile) system (more commonly known as windrow composting). The term ‘turned’ applies to the method used for aeration. In essence, it consists of tearing down a pile and reconstructing it (Diaz *et al.* 1993).

Much research into all aspects of traditional composting is on-going (e.g. De Bertoldi 1996; Stentiford 1997; Bidlingmaier *et al.* 1999) and shall be drawn upon in subsequent Chapters.

1.3.3 The vermicomposting process

Vermicomposting, as with traditional composting methods, has a long research history that can be traced back to the work of Charles Darwin. Vermicomposting stems from 'earthworm farming' (vermiculture) set up in the 1930 and 40s to rear large quantities of fishing bait. Pyramid selling of 'vermiculture kits' and the poor performance of the earthworm species commonly reared as bait (*Eisenia fetida*), led to the exploration of alternative uses for these worms (q.v. Knight 1989).

The natural occurrence of earthworms at sewage works led to research into the use of worms for the treatment of sewage sludge (Hartenstein 1978). This work was then expanded on in subsequent conference proceedings (Applehof 1981). However, much of the research that followed remained focussed largely on the production of earthworms (Tomati & Grappelli 1984; Lofs-Holmin 1985).

Research then shifted to investigate the use of earthworms to treat animal, vegetable and industrial wastes (Edwards & Neuhauser 1988). There was gaining recognition of the economic value of waste treatment and the vermicomposted products (Foote & Fieldson 1982; Fieldson *et al.* 1985; Fieldson 1988). Moreover, the composted products of waste vermicomposting were increasingly investigated for their horticultural value (e.g. Edwards & Burrows 1988).

A large research group based in the UK investigated the use of specially designed beds (large shallow containers) with automated methods of waste application for vermicomposting (Edwards 1988). The most successful type of earthworm species employed for vermicomposting, as well as their nutritional, biological and environmental requirements were also investigated.

Vermicomposting, as with traditional composting involves a range of closely controlled parameters, e.g., temperature, moisture, nutritional composition of the waste. Also many other physicochemical and biological factors must be controlled, specific to the growth, survival, and reproduction of earthworms (Hartenstein *et al.* 1979b; Neuhauser *et al.* 1980a; Kaplan *et al.* 1980b; Hartenstein 1982).

These factors relate predominantly to the production of earthworms, rather than the decomposition and treatment of wastes (*op. cit.* 1988).

Vermicomposting, unlike traditional composting, operates under lower temperature (20-25°C) and higher moisture (70-80%) regimes. Aeration is achieved through the burrowing activity of the earthworms, as well as the ingestion and egestion of waste material as casts. Smaller

amounts of waste are processed in a more continuous process, adding fresh layers of waste once previous applications have been processed (Edwards 1988).

Although it is well established that earthworms increase rates of decomposition (e.g. Loehr *et al.* 1984), the stabilisation of waste has not been well studied and many fundamental factors need to be evaluated to assure the technical and economic success of such processes (Edwards & Bohlen 1996).

Much research into all aspects of vermicomposting is on-going (e.g. Kretzschmar 1992; Edwards & Bohlen 1996; Edwards 1997; Edwards 1998) and shall be drawn upon in subsequent Chapters.

1.3.4 The effect of windrow-composting and vermicomposting on organic waste

Mass/volume reduction

Both windrow-composting and vermicomposting processes result in organic matter loss via microbial metabolism into CO₂ and H₂O. Windrow-composting, an exothermic process, can also result in considerable moisture loss through evaporation (Stentiford 1996). Low temperatures associated with vermicomposting operations (20-25°C) result in lower rates of desiccation. Volume reduction may also occur due to structural changes within composting material. During composting material is broken down into smaller particles (Gray *et al.* 1971), and may become more compact. Vermicomposting also results in the reduction of particle size as small amounts of waste are ingested by earthworms before being egested as earthworm faeces - casts (Mitchell 1978).

These processes lead to reductions in the mass and volume of material remaining for disposal; reducing final disposal costs. The potential for pollution after disposal is also reduced, e.g., lower moisture levels within composted materials reduce the risk of leachate production within landfills (White *et al.* 1995).

Stabilisation

Stability determines the extent to which readily biodegradable organic matter in organic waste has been decomposed. There are a number of methods which can be used to measure stability, such as, microbial respiration (i.e., O₂ uptake and CO₂ production), temperature, and physicochemical analyses including organic matter loss (Lasaridi & Stentiford 1997). These analyses can be used to monitor and compare performance in different composting systems (Stentiford 1993).

The stabilisation of various organic wastes has been investigated using traditional composting techniques (e.g., Diaz *et al.* 1993 and De Bertoldi *et al.* 1996).

However, stabilisation processes during vermicomposting, vermistabilization (Loehr *et al.* 1984), have received little attention. Some studies on how earthworms affect the decomposition of organic wastes include: Pincince *et al.* (1981) and Neuhauser *et al.* (1988) (sewage sludge); Frederickson *et al.* (1997) (green waste); and Vincelas-Akpa & Loquet (1997) (woody, lignocellulosic waste).

Stabilised composted products consist of recalcitrant organic matter, some originating in the feedstock (e.g., lignocelluloses and lignin), as well as new, less biodegradable forms of organic material formed during composting (e.g., microbial remains, and humus). Composted organic materials can also be rendered more stable due to a lowering of their moisture content.

In terms of disposal practices, stabilised materials are less likely to produce harmful leachate, e.g., phytotoxicity and BOD of green waste leachate decreases with composting duration (Frederickson 1997). Composted organic matter is less likely to emit greenhouse gases with high global warming potential (e.g., methane) if disposed of at landfill (Hindle & McDougall 1997). It is also less likely to emit odours (Miller 1993).

In terms of composted product quality, stability is closely associated with reduced phytotoxicity (Zucconi *et al.* 1985), and increased availability of plant nutrients, e.g., nitrate (Finstein & Miller 1985). Though maximum stabilisation during processing is beneficial prior to waste disposal, if the objective is to produce a quality composted product, higher levels of organic matter and microbial activity may be desired, e.g., as in soil amendments (Sela *et al.* 1998).

Sanitation/pathogen reduction

Both windrow-composting and vermicomposting have been shown to reduce the number of pathogenic organisms in organic wastes. High temperatures during the early stages of windrow-composting can destroy pathogenic micro-organisms (Gray *et al.* 1971). To achieve pathogen destruction during windrow-composting the USA's EPA recommend temperatures of $\geq 55^{\circ}\text{C}$ for 15 days (Riggle & Holmes 1994). High temperatures associated with windrow-composting can also destroy the viability of weed seeds (Tomkins *et al.* 1998).

Pathogen destruction during vermicomposting, a low temperature process ($\sim 20^{\circ}\text{C}$), is less certain. Pre-composting material prior to vermicomposting has been suggested as a means of ensuring pathogen destruction (Frederickson *et al.* 1997). However, it is suggested pathogens are killed, not only by temperature, but also by the formation of antibiotic substances produced during composting (Gray *et al.* 1971). Studies have suggested earthworms can reduce numbers of certain human pathogens in wastes during vermicomposting (Mitchell 1978; Murray & Hinkley 1992).

Adequate sanitation during organic waste treatment produces a material less likely to pose a threat to human health during handling and disposal. Sanitation is also essential to producing a good quality composted product, i.e., free from plant pathogens and viable weed seed. Inhibition of plant pathogens by some composted materials can continue when used as horticultural agents (Hoitink *et al.* 1997).

Saleable products from windrow-composting and vermicomposting

The main economic benefits of vermicomposting are identified as the production of compost, reductions in waste disposal costs; and the production of earthworms (Fieldson 1988).

A similar analysis is held for windrow-composting, excluding earthworm production (Renkow & Rubin 1998).

Which of these elements are considered the main goal of a particular composting system will depend upon technical and economic aspects of each waste scenario. For example, the characteristics of the waste under treatment will effect the quality of the composted product (Haug 1996) and, in the case of vermicomposting, the production of earthworms (Neuhauser *et al.* 1980a).

Composted organic matter

Windrow-composting and vermicomposting both produce composted products that can have applications in agriculture or horticulture. The potential utilisation of such products has been widely investigated: vermicompost - Edwards & Burrows (1988), Handrek (1986) and Scott (1988); and compost - Roe *et al.* (1997a/b) and Burger *et al.* (1997).

However, vermicomposting and windrow-composting operate under very different conditions, sometimes producing very different products (Dominguez *et al.* 1997). Few comparative studies of vermicomposts and traditional composts have been conducted and often there are disparities in the waste materials and/or composting systems/conditions employed, e.g., Casalicchio & Graziano (1987), Haimi & Huhta (1987) and Subler *et al.* (1998).

Earthworm biomass production (vermiculture)

Historically, studies on vermicomposting have focused on earthworm biomass production (vermiculture); with many investigations into physicochemical and biotic factors affecting earthworm growth and fecundity (Lofs-Holmin 1985). Traditionally earthworms were bred for fish bait in a wide range of organic wastes (Edwards 1998). This process, commonly known as 'earthworm farming' was considered to have great economic potential (Gaddie & Douglas 1975; Barrett 1976). This led to many studies into the feasibility of earthworm farming (e.g., Pincince *et al.* 1981; Tomati & Grappelli 1984).

The nutritional value of earthworms has long been established (Lawrence & Millar 1945; Sabine 1978), and studies have been conducted utilising earthworms as animal feed (Fisher 1988; Tacon *et al.* 1983). It has been suggested earthworms could even provide a food source for humans (McInroy 1971). In addition, the production of soil dwelling earthworms for land reclamation and amelioration has been investigated (Butt 1993).

Pincince *et al.* (1981) suggested earthworm species used in vermicomposting were not favoured by anglers and not suited to the conditions of agricultural soils. The utilisation of earthworms as a food source could also be restricted by health considerations, e.g., the accumulation of toxic substances in earthworm biomass; and earthworms as vectors of pathogens (*op. cit.* 1981).

Economic studies of earthworm farming suggests the largest economic potential lies within the vermicomposted organic matter, not the earthworms (Foote & Fieldson 1982); although, worm production could have better prospects under the right circumstances (Fieldson *et al.* 1985).

Fieldson (1988) suggests areas of vermicomposting research that need to be addressed, including the rate at which worms convert different types of wastes; which may have a major effect on profitability.

This thesis therefore includes investigations into the rate at which worms convert waste paper sludge into compost.

Direct environmental impacts of composting

Environmental benefits of biological waste treatment depend on numerous factors: the environmental impact of the current method of waste treatment and that of the method under investigation.

Each case must be examined individually using data from all aspects of the waste treatment process. Holistic methods, e.g., environmental impact assessment (Petts & Eduljee 1994) or life cycle assessment (White *et al.* 1995), have been proposed as means of assessing the environmental and economic implications of a particular waste treatment system.

A detailed discussion of such techniques is beyond the scope of this study, although trends and principles of such methods are alluded to in other sections.

Composting organic waste can directly benefit the environment by reducing the environmental cost of current disposal routes, e.g., lowering the potential for greenhouse gas emissions and ground water pollution from landfills (Hindle & McDougall 1997).

Landfill gas (approximately 1/1 v/v CO₂ and CH₄) can be collected and used as a fuel for energy production. However, only a third of active landfills taking 'significant' quantities of biodegradable wastes in England and Wales possess gas control equipment (ETSU 1998). Landfill gas is the largest man-made source of methane, thought to contribute 31% of the UK's total methane emissions (Hindle & McDougall 1997), although recent research suggests that this may be an overestimation (ENDS Daily 1999).

Composted products can also provide environmental benefits, e.g., returning organic matter to the soil (Steffen 1979) and reducing the need to use scarce natural resources such as peat (Knight 1991).

However, composting also carries environmental costs, producing potentially harmful emissions. The proximate environmental effects of composting plants are increasingly under investigation (e.g. Wheeler 1997 and Fischer 1996).

Emissions receiving particular attention include: gaseous emissions - odours (e.g., Bidlingmaier 1996), greenhouse gases (e.g., Kuroda *et al.* 1996) and volatile organic compounds (e.g., Brown *et al.* 1997); liquid emissions - leachate (e.g., Frederickson 1997); micro-organism emissions - bioaerosols (e.g., Gilbert & Ward 1998); and particulate emissions - dust (e.g., Frazer *et al.* 1993).

1.3.5 Comparative research into windrow composting and vermicomposting

It can be clearly seen that treatment of organic wastes by traditional composting and vermicomposting share many similar environmental and economic benefits; with the exception of earthworm production.

Although, it is recognised that these composting methods operate under very different conditions and can produce very different composted products (Dominguez *et al.* 1997). However, few comparative studies of these processes have been conducted.

Comparisons between traditional composting and vermicomposting focus mostly upon the final composted products (Haimi & Huhta 1987; Subler *et al.* 1998).

There are few comparative investigations into differences between physicochemical transformations during vermicomposting and traditional windrow composting.

Vinceslas-Akpa & Loquet (1997) examined organic matter loss and organic matter transformations during the vermicomposting and composting of maple waste, under small scale laboratory conditions. However, the waste investigated was possibly not suitable for vermicomposting. The composting aspect of their trial consisted of the incubation of the waste substrate under environmental conditions similar to those used during vermicomposting

but without worms. This is not comparable to thermophilic conditions normally associated with large-scale windrow composting systems (see above).

Gellens & Verstraete (1995) evaluated a large-scale batch vermicomposting system, processing pre-composted VFGP (vegetable food garden paper) waste mixed with non-recyclable paper. The investigation focused upon the processing capacity of the system rather than monitoring changes within organic matter during processing.

There is a clear gap in comparative studies of traditional windrow composting and vermicomposting, and this provides the predominant theme of this thesis.

1.4 biological organic waste treatment within the UK paper industry

1.4.1 Introduction

UK industry provides large opportunities for environmentally sustainable waste treatment practices that may provide economic savings and improvements in environmental protection; advocated by the UK's ETBPP (section 1.2.4).

Industrial wastes are considered to be more consistent than other waste sources; 'wastes arising in these less-visible [industrial] streams are generally more homogeneous, cleaner, and more accessible than domestic waste' (DETR 1998a). Industry, therefore, may provide feedstocks capable of producing more consistent composted products; starting materials largely determine the quality of composted products (Lopez-Real & Merillot 1996).

The potential use of biotechnology, including composting, in organic waste treatment within industry is recognised by the UK's Department of Trade and Industry (DTI):

'Bioprocesses are already well established in the treatment of municipal and industrial waste-waters. Naturally-occurring microbes are responsible for the breakdown of the majority of organic substances released into our environment...Bioprocesses are thus likely to provide environmentally-friendly routes to waste treatment. In many circumstances they are also the most cost effective...offering low operating costs compared to processes which require large amounts of chemicals or energy' (DTI 1997a).

The DTI extends the opportunities for using biotechnology to the UK paper and pulp industry, dealing with organic waste, such as, sludges originating from paper mill waste water treatment plants:

'...the [paper and pulp] industry is subject to strict environmental controls. Biological waste treatment is already an accepted practice in the paper industry. This role will continue with tightening environmental legislation. Most of the wastes do not present unique problems and there will be opportunities to import technologies from other industries...Outside the waste treatment area the benefits for biotechnology will be relatively minor...It is important UK companies become aware of and are in a position to exploit any new technologies that are developed if they are to remain competitive' (DTI 1997b).

The effects of environmental regulation on industry, 'the greening of industry', are generally considered to have a positive impact on innovation and competitiveness at the company level, although, industry-level studies give more divergent results (Clarke & Georg 1995).

1.4.2 Environmental progress within the pulp and paper industry

European pulp and paper manufacture appears to be one industry greatly affected by trends in waste legislation. This is emphasised by the industry's commitment to the concept of, 'a closed cycle mill - one entirely without emissions' (CEPI 1998).

Paper manufacture is traditionally a process that requires large amounts of raw materials, water, energy and chemicals; resulting in large levels of waste and emissions. Technical innovation in many areas of manufacture have resulted in environmental improvements. The pulp & paper industry in Europe has witnessed a number of large reductions in emissions, e.g., 90% reduction in chlorinated organic compound emissions since the mid-1980s, 90% reduction in sulphur dioxide compound emissions since the 1970s, 85% reduction in oxygen consuming organic compounds in waste waters; and reductions in water usage and energy consumption (CEPI 1998).

The paper industry has also experienced changes in raw material utilisation (CEPI 1998). There is a long tradition of waste paper recovery within the industry, which has increased during the 1990s. European (EU, Norway & Switzerland) recovery rates have increased from 39.3% in 1991 (CEPI 1998) to 48.9% in 1997 (CEPI 1998), an increase of over 10 million tonnes of waste paper used in the production of new paper products (CEPI 1998). This trend is encouraged by legislative pressures (e.g., EU Directive on Packaging Waste), and a voluntary commitment to an EC target set up in 1993, i.e. 50% recycling/re-use by the year 2000 for all paper consumed in the EU (CEPI 1998).

In the UK the amount of waste paper used as raw material in paper manufacture doubled between 1987 and 1997. Waste paper pulp accounted for approximately 63% of the raw materials used in paper manufacture in the UK during 1997 (PFGB 1998). This is encouraged by legislation, e.g., UK Packaging Waste Obligations Regulations (DoE 1997), and voluntary initiatives, e.g., UK Newspaper Publishers Association working group, set up to consider how recycled fibre content in papers can be increased (DETR 1998b).

1.4.3 Waste paper sludge - a candidate for biological waste treatment

The environmental impacts of waste paper recycling are complex. In a life-cycle analysis of paper recycling, Virtanen & Nilsson (1993) suggested recycling could lead to unexpected effects, offsetting expected benefits. In the investigation of scenarios with increased paper recycling, one effect was an increase in total suspended solids (TSS) and biological oxygen demand (BOD) in waste waters.

Waste paper fibre compared to virgin fibre is less efficient as a raw material. Recycled fibres become weaker with every cycle and lose their paper making qualities (CEPI 1998); waste paper also contains varying amounts fillers and coatings which must be removed before the fibre can be used. This may explain the high waste paper utilisation rates of certain recycled paper products, e.g., the production of case materials in the UK has a recovered paper utilisation rate of greater than 100%, giving a recovered paper fibre content of approximately 85% (PFGB 1998).

Despite the potential for increases in the suspended solids of paper mill waste waters, the BOD₅ (biological oxygen demand over a 5 day incubation period) levels of UK paper mill waste water emissions have decreased greatly, from 12 Kg/tonne in 1990 to ~1.5 Kg/tonne in 1995 (CEPI 1998). This has been achieved through increased removal of organic matter during waste water treatment, driven by increasingly strict waste water discharge regulations, e.g., EC Directive on Urban waste water (EC 1991b). A result of elevated levels of TSS in paper mill waste waters combined with increased organic matter removal has been increased production of paper mill waste water treatment sludges - waste paper sludge (WPS).

Approximately 700, 000 tonnes (wet weight) of WPS were produced in the UK in 1994, which is predicted to increase to 1,000, 000 tonnes by the year 2000 (Sesay *et al.* 1997). It is estimated approximately half of the UK WPS production originates from recycled paper mill pulping and de-inking processes (Sesay *et al.* 1997). The production of WPS may be further augmented by increases in paper and board production. UK paper production has grown from 4.2 million tonnes in 1987 to 6.5 million tonnes in 1997, largely through increased paper recycling methods of production (PFGB 1998).

Disposal of waste paper sludge (WPS) is an increasing problem for many UK paper mills. Not only has waste legislation indirectly increased WPS production through tighter regulations on water emission, and encouraging increased paper recycling, but it is also imposing increasing costs and other restrictions upon its disposal.

The value in finding alternative uses and/or treatments for WPS have led to numerous investigations, e.g., incineration (Halonen *et al.* 1993); land application (Tripepi *et al.* 1996); composting (Sesay *et al.* 1997); vermicomposting (Elvira *et al.* 1997); vermiculture (Butt 1993 and Fayolle *et al.* 1997); horticulture (Bellamy *et al.* 1995); microbial-protein production (Kannan *et al.* 1990); ethanol production (Lark *et al.* 1997); anaerobic digestion (Ratnieks & Gaylarde 1997).

Much of this research shows there is potential for the treatment of WPS using biological processes. However, it also demonstrates the variety of WPS types produced by different paper mill waste water treatment processes; and the need to find the most appropriate

substrate for each treatment process and its associated objective, e.g., compost production or earthworm production.

1.4.4 Windrow composting and vermicomposting of waste paper sludge

Traditional composting systems have been conducted for the treatment of various waste paper sludges with and without nutritional or structural amendments (Campbell *et al.* 1991; Brouillette *et al.* 1996; Sesay *et al.* 1997; Jackson & Line 1997a, b, c & 1998; Hackett *et al.* 1999).

Vermicomposting systems have also been demonstrated as feasible methods for treating waste paper sludge (Elvira *et al.* 1995a, b, 1997 & 1998).

Numerous studies into the horticultural and agricultural value of composted and non-composted waste paper sludge are documented (Bellamy *et al.* 1995; Campbell *et al.* 1995; Chong & Cline 1993 & 1994; Chong & Hamersma 1996; Tripepi *et al.* 1996).

This body of research was drawn upon to further investigate both the vermicomposting and windrow composting of waste paper sludge. The apparent suitability of this waste to both processes was also investigated with a view to using its as a substrate for a balanced comparison of both processes.

1.5 Integrated waste management

Designing a sustainable waste management system implementing all the principles promoted within current waste legislation, e.g., waste hierarchy, producer responsibility, BPEO, and proximity principle (section 1.2), is a complex problem which necessitates the utilisation of various waste management techniques. White *et al.* (1995) suggest sustainable waste management should be achieved using an integrated system of various waste treatment processes, handling all types of waste, from all sources. This trend is reflected in the UK waste management industry, 'changing from small firms to large integrated companies offering a range of waste management services' (DETR 1998a). The European paper and pulp industry recognises a need for using an integrated system of techniques, as they move toward implementing the EU Directive on Integrated Pollution Prevention and Control (CEPI 1998).

When designing an integrated waste management system, White *et al.* (1995) advocate using an holistic modelling tool, 'life-cycle assessment' (LCA), to examine the whole life-cycle of waste, both economically and environmentally, from the 'cradle' (when material is produced without value or a product becomes valueless) to the 'grave' (conversion into a valuable product; or disposal). The UK Environment Agency supports this approach, and is developing LCA into a new strategic waste management tool, to be made available as a software package in 1999 (Environment Agency 1998). LCA for solid waste treatment, or for any other product or service, is mainly used in a comparative way (White *et al.* 1995). Economic and environmental data from all aspects of waste treatment or disposal processes can be used to find the 'best option', a balance between environmental benefit and economic cost, e.g. the BPEO (section 1.2.4).

It is generally accepted that biological waste treatment has an important role within integrated waste management systems. However, biological treatment will not be able to deal with all wastes, and will need to be compared with other waste treatment/disposal options (White 1996). Studies into the utilisation of LCA and BPEO as methods for assessing biological and non-biological organic waste treatments have been conducted (e.g., Barton 1997 and Sonesson 1997). Both Barton and Sonesson outline the complexities of such methods and recognise a lack of reliable data regarding processes and waste composition (Sonesson *et al.* 1997); for example, Barton (1997) outlines a lack of consistent design, operation, performance, and product standards in composting. In a solely economic study of vermicomposting, Fieldson (1988) recognised gaps in the knowledge of the vermicomposting process, such as processing rates.

The need for integrated waste management systems and a lack of comparative data concerning treatment options, again show the benefits of the comparative research such as will be conducted herein.

1.6 Main aims of research

This study attempts to provide data, on the technical feasibility of processing waste paper sludge waste using windrow-composting and vermicomposting systems; with an emphasis on comparing aspects of process and product:

1. Investigate the technical potential of windrow-composting to process waste paper sludge in terms of organic decomposition.
2. Investigate the technical potential of vermicomposting to process waste paper sludge in terms of earthworm growth and mortality.
3. To compare windrow-composting and vermicomposting of a waste paper sludge in terms of decomposition rates, and quality of composted products.

1.7 Thesis outline

Chapter 2 - Windrow composting of primary and secondary waste paper sludge (WPS)

Three large scale experiments investigate the potential of windrow-composting techniques to decompose WPS. Two types of waste paper sludge were examined, derived from different stages of paper mill waste water treatment plants, and originating from two paper recycling mills. Treatments aimed at improving conditions for composting were tested: addition of mineral nutrients; addition of an organic bulking agent; and co-composting a mixture of waste paper sludges.

Chapter 3 - Earthworm population sustainability and growth

The potential of vermicomposting to process WPS was explored. A suitable WPS was identified and its nutritional value in maintaining a population of earthworms (*Dendrobaena veneta*) was investigated. A range of earthworm stocking densities was employed to investigate the optimisation of WPS processing and decomposition, while maintaining adequate earthworm growth and survival.

Chapters 4 Large scale composting of a commingled primary and secondary waste paper sludge using an open-air, mechanically turned windrow system.

An investigation into large-scale windrow-composting of WPS was investigated. The WPS employed was that found to be suitable for vermicomposting (Chapter 3) and possessed the physicochemical characteristics required for windrow composting. Two windrow-composting parameters were investigated to optimise WPS stabilisation: the use of a bulking agent and a frequent turning regime.

Chapter 5 - Large scale vermicomposting of a commingled primary and secondary waste paper sludge using a modular batch vermicomposting system.

An investigation into large-scale vermicomposting of WPS was conducted. The WPS employed was that found to be suitable for vermicomposting (Chapter 3). Two vermicomposting systems were investigated to optimise WPS stabilisation: the introduction of earthworms to a large single-batch of WPS and the introduction of smaller, more frequent batches for shorter time-periods.

Chapter 6 - Windrow-composted and vermicomposted waste paper sludge as components of plant growth media.

An investigation into composted-WPS products from windrow-composting and vermicomposting processes was conducted. Composted products from optimised vermicomposting and windrow composting systems were utilised in the cultivation of a plant species suitable for horticultural assessment.

Chapter 7 - A comparison of mechanically turned windrow composting and vermicomposting in the stabilisation of WPS into a horticultural product.

A direct comparison of windrow composting and vermicomposting in terms of processing rates and the physicochemical properties of their respective products is conducted using the data obtained in Chapter 4, 5 and 6. To conduct as balanced a comparison as possible, results were taken from systems proved to be optimal in each case.

Chapter 8 - Main findings and suggestions for further research

All experimental findings are discussed. Possible implications of these findings with regard to technical, economic and environmental potential of composting and vermicomposting in the paper manufacturing industry are examined. Potential areas further investigation are outlined.

2. Windrow composting of primary and secondary waste paper sludge (WPS)

2.1 Introduction

As discussed in Chapter 1, there is a pressing need to develop sustainable, biologically-based processes, such as composting, for treating the highly polluting waste sludges produced by the paper making industry. However, the physicochemical properties of waste paper sludges (WPS) vary greatly, depending upon the raw material used in the paper making process, and the waste water treatment process employed (Webb 1994). As a consequence of this, Jackson & Line (1997a, b) have suggested the development of sludge-specific methods of composting. The research programme described in this section will attempt to devise and evaluate windrow composting practices specifically tailored to the characteristics of sludges, typically produced by the UK paper making industry.

A particular feature of the programme was to devise approaches to composting which could be adopted relatively easily by paper making plants, which are often based in remote locations. While it is acknowledged that much useful research has been undertaken on composting WPS sludges using static aerated piles and by supplementing with other wastes (Sesay *et al.* 1997), the approach to this work was to minimise intervention. Hence, much emphasis has been placed throughout this work on investigating the effective stabilisation of sludges using the minimum amendment of sludges and the least complex technology. It is envisaged that this approach will relate more closely to the financial and technical needs of the paper making industry in the UK than technologically complex solutions and the requirement to import waste supplements as feedstock for composting.

This Chapter has three main aims and these are:

1. to identify typical waste paper sludges which should be theoretically amenable to windrow composting
2. to determine the relevant physicochemical properties of the selected sludges and to suggest ways of ameliorating sludge properties to promote effective composting
3. to monitor key parameters during composting and to evaluate both the amelioration methods and the potential of composting to stabilise each sludge

There have been numerous studies into the application of composting techniques in treating a variety of waste paper sludges and these will be discussed in later sections: Mick *et al.* 1982;

Carter 1983; Valente *et al.* 1987; Campbell *et al.* 1991 & 1995; Chong & Cline 1994; Line 1995; Brouillette *et al.* 1996; Sesay *et al.* 1997. Some commercial success using such processes has also been achieved (Smyser 1982).

This study investigates two WPS types, both products of paper recycling mills, one resulting from primary waste water treatment and one resulting from secondary waste water treatment. Differing physicochemical characteristics of the WPSs were analysed and their suitability for mechanically turned windrow composting was investigated.

Primary (1°) waste water treatment (Appendix 1) typically produces WPSs consisting of unrecoverable fibres, fillers and absorbed/adsorbed chemicals including coagulants; screw or belt pressed to a moisture content of 40–50% (Webb 1994). 1° WPS contains low levels of mineral–nutrients giving high carbon/nutrient ratios. The organic matter content of which consists mainly of lignocellulosic fibres and hemicellulose.

Secondary/biological (2°) waste water treatments produce WPS much higher in macro–nutrients such as nitrogen (N), phosphorus (P) and potassium (K), which are added as part of the waste water treatment process (Appendix 1). In the case of secondary (2°) WPS, paper fibres have been broken down into more readily degradable organics, *e.g.* sugars, amino–acids and microbial biomass (Bellamy *et al.* 1995). The sludge consists of flocculated organic and inorganic matter, forming a gelatinous precipitate which is resistant to de–watering. Thus, secondary (2°) WPS possess high moisture contents of 70–80% (Webb 1994).

Experimental programme

An experimental, large-scale composting programme was devised to evaluate the effect of three treatments (ie three sludge amendments) on the composting of the primary and secondary sludges under study. A particular feature of the programme was the additional use of control windrows containing unamended sludges, which were employed to evaluate the effects of the treatments. As described above, one of the key drivers in the research was minimum intervention and in particular, reducing dependence on importing supplementary wastes, as per the proximity principle. Hence, amendment of sludges was kept to a minimum.

Experiment 1 – The effect of nutrient addition to primary sludge

From an analysis of Table 1, Primary WPS, possessing low levels of macro–nutrients (N, P, K), does not appear to have appropriate carbon/nutrient ratios for efficient composting. This experiment investigates the use of inorganic fertiliser as a nutrient source when composting 1°WPS. It was hypothesised that the addition of fertiliser will lower carbon/nutrient ratios to levels more favourable to microbial activity, accelerating 1°WPS decomposition.

Experiment 2 – The effect of bulking agent addition to secondary sludge

With high levels of macro-nutrients (N, P, K), biological activity and moisture content, 2° WPS is a difficult waste to handle with a potentially high environmental impact. As for primary sludge, WPS does not appear to have the correct nutrient balance for efficient composting. The secondary sludge's poor physical structure, high bulk density and high moisture content would appear to render the sludge unsuitable for composting without amendment. It also has the potential to produce high levels of odour and leachate. This experiment investigates the use of a bulking-agent when composting 2°WPS to improve the porosity of the mix and as a carbon-source. It was hypothesised that the addition of straw would modify carbon/nutrient ratios, moisture content, and windrow structure; thereby allowing composting to take place and accelerating 2°WPS decomposition. Straw is commonly used as a conditioning agent in sewage-sludge composting (Lopez-Real *et al.* 1989). Sewage sludge has similar physicochemical characteristics to 2°WPS, a product of similar waste water treatment processes.

Experiment 3 – The effect of combining primary and secondary sludges

It has already been highlighted that for effective composting, unamended primary and secondary WPSs are deficient in a number of key physicochemical characteristics. In this experiment, it was hypothesised that mixing primary and secondary WPSs would produce a feedstock with carbon/nutrient ratios, moisture content, and physical structure adequate for efficient composting. The mixing of 1°WPS and 2°WPS to correct nutrient imbalances prior to land application has already been suggested (Pridham & Cline 1988).

2.2 Materials and methods

2.2.1 Formulation of windrow treatments

Table 1. – Physicochemical estimations

Physicochemical character	Feedstock		Treatment		Physicochemical estimations of Treated windrows			
					Expt. 1	Expt. 2	Expt. 3	
	1°WPS	2°WPS	Straw	Fertiliser solution*	1°WPS 2300 l fertiliser	2°WPS Straw 1:1 vol†	1°WPS 2°WPS 2:1 vol	1°+2° WPS 1200 l fertiliser
Bulk density (g l ⁻¹)	556	964	63	1000	–	–	–	–
Total solids (%)	64.8	27.0	78.0	0.024	42	35	45	43
TOC (%DM)	23.3	23.5	47.2	6.6	23.0	27.3	23.4	23.0
Total P (%DM)	0.05	1.33	0.30	15.6	0.36	1.17	0.39	0.66
Total K (%DM)	0.14	0.35	2.3	14.6	0.45	0.66	0.20	0.47
TKN (%DM)	0.35	2.62	0.5	14.9	0.68	2.28	0.95	1.23
C/N ratio	66.9	9.0	94.4	0.4	34.1	11.9	24.6	22.9

* Fertiliser solution consisted of a solution of 1:1:1 (N:P:K): 40 kg of Urea [CO(NH₂)₂] (46.7% N), 40 kg of phosphate [P₂O₅] (43.7% P), 40 kg of potassium sulphate [K₂SO₄] (44.8%K) in 5000 litres of water.
† It was hypothesised that this was the minimum volume of straw required to provide an adequate moisture level and windrow structure.

The total dry mass (TDM) of each component in a mixture was estimated using bulk density, total solids content, and volume:

TDM = Total Solids × Bulk Density × Volume (1)

The Final moisture or nutrient levels of mixtures (X_{mix}) were calculated from TDM and the initial moisture and nutrient values (X) of each component:

X_{mix} = (X₁ × TDM₁) + (X₂ × TDM₂) / TDM₁ + TDM₂ (2)

2.2.2 Feedstock preparation and windrow formation

Experiment 1

A feedstock of 1°WPS was obtained from Aylesford Newsprint paper–recycling mill, Kent, UK; a product of their primary waste water system (Appendix 1).

1°WPS was formed into two windrows* 7 m x 1 m x 2.5 m (length × height × width) each containing approximately 7 tonnes of WPS. Approximately 2300 litres of fertiliser solution was added to the treated windrow and 2300 litres of water was added to the control windrow.

Experiment 2

A feedstock of 2°WPS was obtained from Townsend-Hook paper recycling mill, Kent, UK; a product of their secondary waste water treatment system (Appendix 1).

2°WPS was formed into two windrows each containing approximately 7 tonnes of WPS. The control consisted of unamended 2°WPS formed into a conical pile, 0.5 m × 5.0 m (height × width). The treated windrow consisted of a mixture† of 2°WPS and barley–straw in a ratio of 1:1 by volume (based on the baled volume of straw); formed into a pile 7 m × 1 m × 2.5 m (length × height × width).

Experiment 3

1°WPS and 2°WPS were mixed in a ratio of 1:1 by volume and formed into two windrows, 7 m × 1 m × 2.5 m (length × height × width); each containing approximately 7 tonnes of WPS. Approximately 1200 litres of fertiliser solution was added to the treated windrow and 1200 litres of water was added to the control.

2.2.3 Process Control

Windrow turning regime

All windrows were turned using a tractor mounted with a loading shovel. During each turning process, all composting feedstock was lifted from the base of the windrow and relocated adjacent to its original position. This operation was conducted until all feedstock had been thoroughly mixed and compaction had been alleviated.

The 12 week experimental period consisted of 8 weeks active composting (*i.e.* with turning) and a 4 week maturation period (without turning). During the first 4 weeks of active composting, windrows were turned twice per week; and during the final 4 weeks of active composting, once per week.

Moisture content

Moisture contents were calculated at every sampling period (section 2.2.4.). Water was added from a bowser using a petrol powered water pump to maintain moisture contents above

* Windrows were constructed using a tractor–mounted loading shovel.

† Mixtures were prepared using a tractor–driven rotavator.

inhibitory levels (50%, Miller 1991). Over saturation of the feedstock was avoided to prevent the formation of anaerobic conditions, and nutrient leaching.

Table 2 – Watering regime

	Experiment 1		Experiment 2		Experiment 3	
Week	Treated	Control	Treated	Control	Treated	Control
0	2300 l NPK	2300 l H ₂ O	–	–	1200 l NPK	1200 l H ₂ O
0.5	500 l H ₂ O	–	–	–	100 l H ₂ O	–
1	–	250 l H ₂ O	–	–	–	–
4	500 l H ₂ O	500 l H ₂ O	500 l H ₂ O	–	500 l H ₂ O	500 l H ₂ O
6	500 l H ₂ O	500 l H ₂ O	500 l H ₂ O	–	500 l H ₂ O	500 l H ₂ O

2.2.4 Process monitoring

Windrow temperature

Mean windrow temperatures were recorded at daily intervals during the first 4 weeks of active composting, and then at two day intervals until the end of the 12 week experimental period. Temperature readings were taken from 10 locations on each windrow: 2 probes at a height of 1 m from the base of the windrow and 3 probes at 0.5 m from the base of the windrow, at depths of 0.30 m and 0.6 m. The positioning of probes was kept consistent between recording periods, and windrows.

Each temperature reading was taken using a 1 m probe attached to a thermometer (TES 1300, type K). The probe was placed into the windrow for 2 minutes, to achieve a stable reading.

Sampling regime

Samples were collected at 0.5 week intervals for the first 2 weeks and then at weeks 3, 4, 6, 8 and 12. Four samples were taken from each windrow at each sampling period. Windrows were marked into four equal sections, and from each quarter 10 x 1 litre portions of the composting material were taken from a depth of approximately 0.3 m from the windrow’s surface and mixed thoroughly. From this a 2 litre sub-sample was taken, placed in hermetically sealed plastic bags and kept in frozen storage until analysis. Samples were taken shortly after turning to increase the homogeneity of the sampled material.

Physicochemical analyses

Moisture content

All samples were analysed for moisture content (% wet mass). Moisture content was estimated as the loss in mass of a 100g portion of sample dried in a desiccating oven at 105 °C for 24 hours.

Total organic matter and carbon content

The total organic matter content (TOM) was determined for all samples. A 100g portion of each sample was dried at 105°C for 24 hours, milled using a knife-mill (Messerm 125H), and passed through a 1 mm sieve. Portions (5g) of the prepared samples were placed in ceramic crucibles, and ignited in a muffle furnace. TOM was calculated as the loss in mass (volatile solids) on ignition at 550°C for 2 hours (FCQAO, 1994).

Total organic carbon (TOC) content was estimated as 55% of the TOM; $TOC = TOM/1.8$ (Adams *et al.*, 1951). This concurs with the range of 45–60% of the TOM (median 52.5%) calculated for recycled WPSs (Bellamy *et al.* 1995); and an average of 51.5% calculated from data for numerous organic waste types (taken from Navarro *et al.* 1993).

*Water extractable nutrients **

Water soluble ammonium (NH_4^+), nitrate (NO_3^-), phosphorus (P), potassium (K) and magnesium (Mg); pH and electrical conductivity (EC) were determined for fresh feedstock and samples taken after 4 and 8 weeks composting.

*Macro nutrient contents **

Total (Kjeldahl) nitrogen (N), total phosphorus (P) and total potassium (K) contents were determined for fresh feedstock and samples taken after 4 and 8 weeks composting.

Mathematical and Statistical analysis

Relative nutrient losses (correcting for dry mass loss)

Nutrient losses were corrected for the concentration effect due to the reduction in organic matter during composting. Ash (%DM) was used as a baseline, assumed to remain at a constant total mass (Stentiford & Pereira Neta 1985).

Using the theory of ash conservation, dry mass (DM) contents can be calculated based on the following equation:

$$DM_t = TOM_t + ASH_0 \quad (3)$$

TOM = total organic matter content after composting duration (t); ASH_0 = initial ash content (adapted from Genevini *et al.* 1996).

Thus nutrient losses may be corrected for loss of DM using the conservation of ash:

* Analyses were conducted by 'Levington Agriculture', Levington Park, Ipswich, Suffolk, UK; using standard ADAS procedures.

$$X\%_{ASH} = \left\{ 1 - \left(\frac{\%X_t \times \%ASH_0}{\%X_0 \times \%ASH_t} \right) \right\} \times 100 \quad (4)$$

X = nutrient; %ASH₀ = initial ash content (%DM); %X₀ = initial nutrient content (%DM); %ASH_t = final ash content (%DM) after composting duration (t); %X_t = final nutrient content (%DM) after composting duration (t) (adapted from Bernal *et al.* 1996).

Non-linear regression of TOM with composting duration

Curves of changing TOM%DM with composting duration were fitted with a one-phase exponential decay model:

$$TOM\%DM = a \times e^{(-\lambda x)} + c \quad (5)$$

a = total loss in TOM (%DM) (initial TOM – c); c = final value of TOM (%DM);

λ = decay constant; Half-life (T) = (ln2)/λ.

Curves were fitted using GraphPad Prism[®] statistical analysis software package (GraphPad Software Inc., CA, USA), running a Marquardt algorithm to minimise the sum-of-squares.

Though usually associated with radioactive decay, the one-phase decay model provided scientific assumptions appropriate for modelling organic matter decomposition, *i.e.* a substance will decay at a decelerating rate before reaching an essentially constant final value. The model was used to estimate values of total loss of TOM (%DM); decay rates (λ & T); and final TOM (%DM).

Statistical differences between treated and control windrows were calculated from the mean and standard error (se) of these variables, using two-tailed Student t-tests; where F-tests suggested standard deviations between populations were possibly unequal (p < 0.05), a Welch's corrected t-test was performed.

Statistical analysis of acidity/alkalinity

Statistical differences in acidity/alkalinity were calculated using [H⁺]; pH was converted into [H⁺] using equation 5:

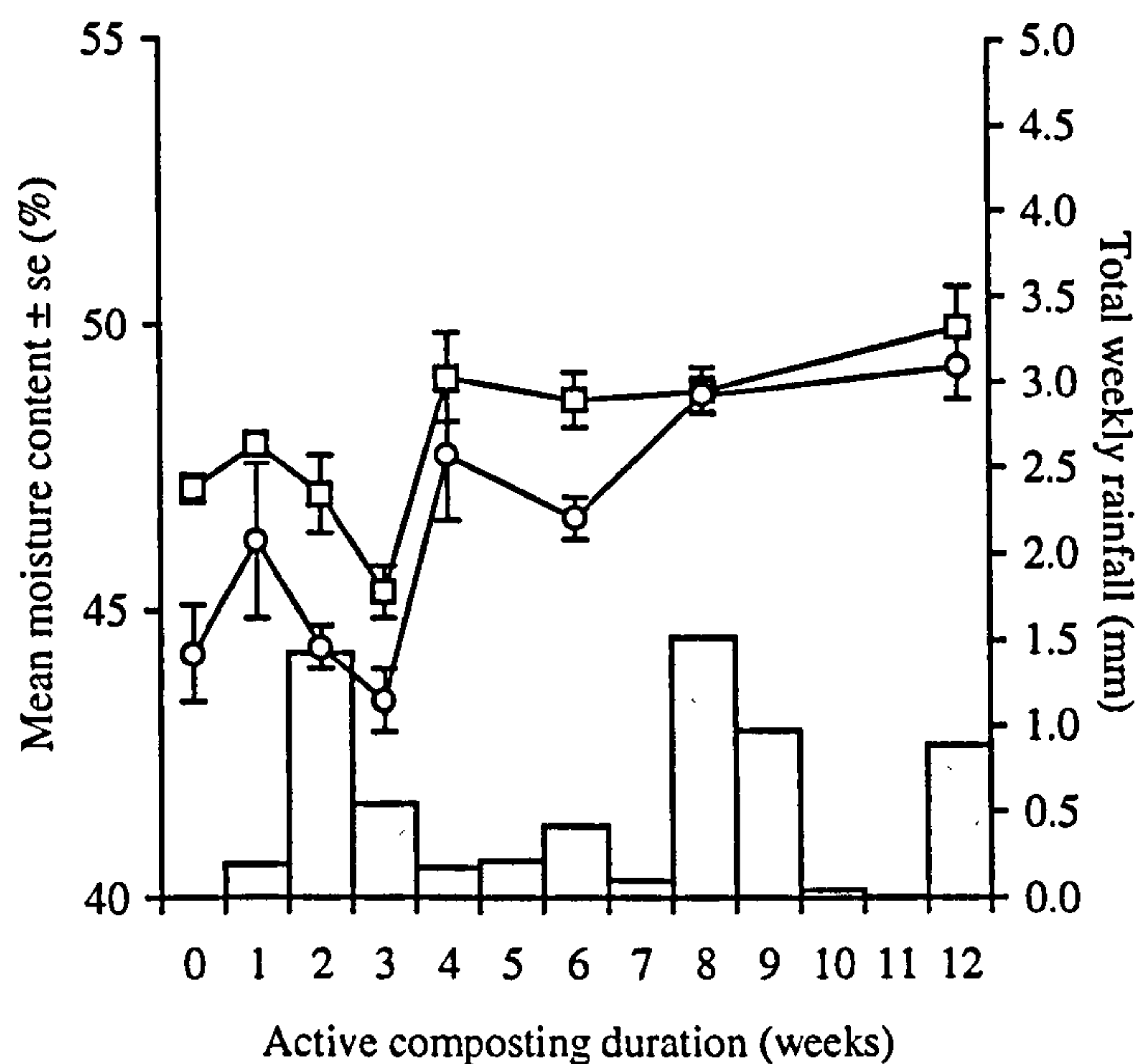
$$[H^+] = 10^{-pH} \quad (5)$$

2.3 Results

2.3.1 Moisture

Experiment 1

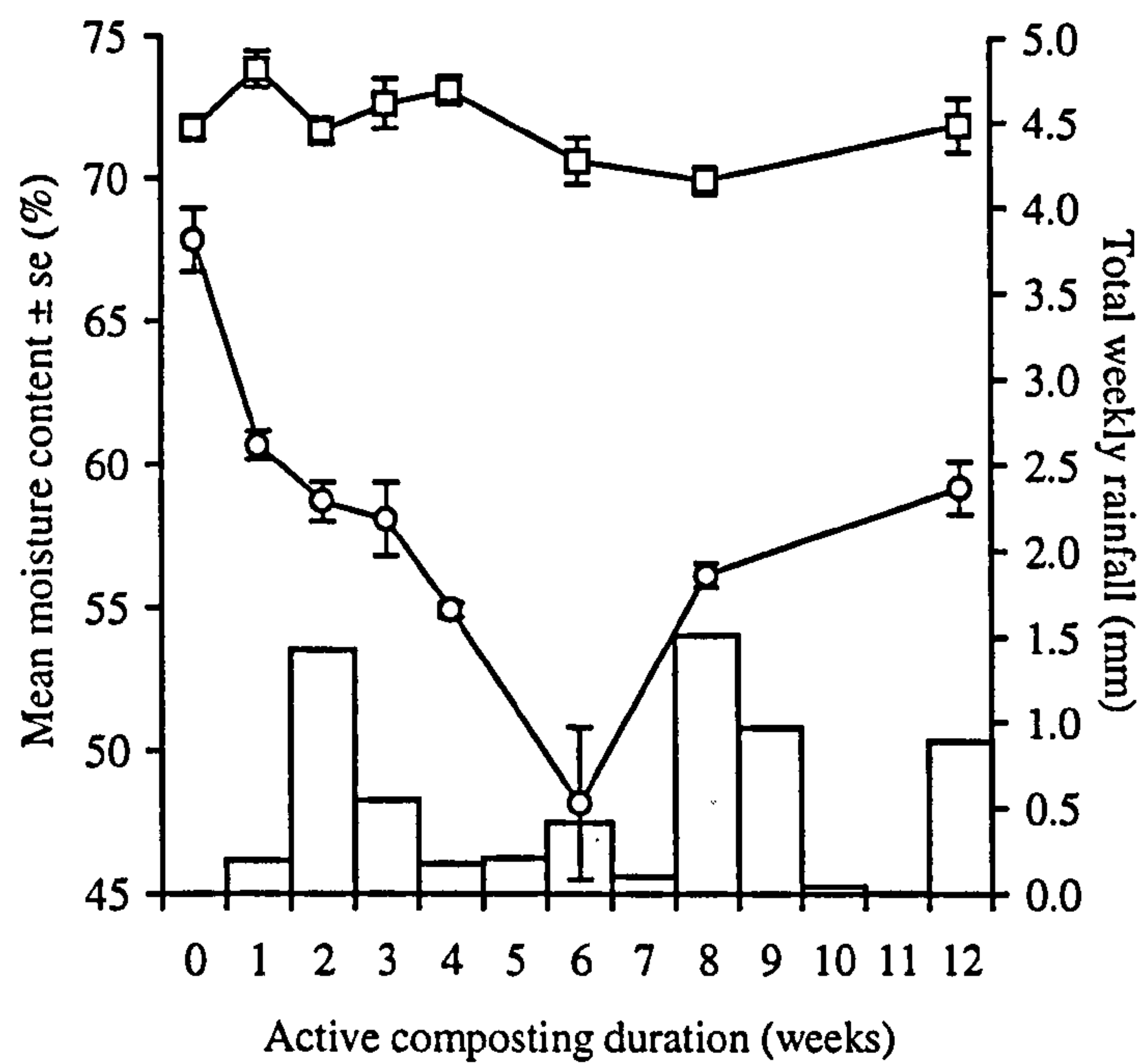
Figure 1– Changes in moisture content (treated windrow –O–; control windrow –□–)



Mean moisture contents of the treated windrow (treated) and the control windrow (control) were initially 44 and 47% respectively, despite adding 2300 litres of liquid fertiliser (treated) and water (control) at the start of composting (section 2.2.3). During the 12 week experimental period mean moisture contents were maintained at between 43–49% for the treated windrow and 45–51% for the control. The mean moisture content of the treated windrow was significantly below the control for the first 3 weeks of composting ($p < 0.05$), with a maximum difference of 4.5%.

Experiment 2

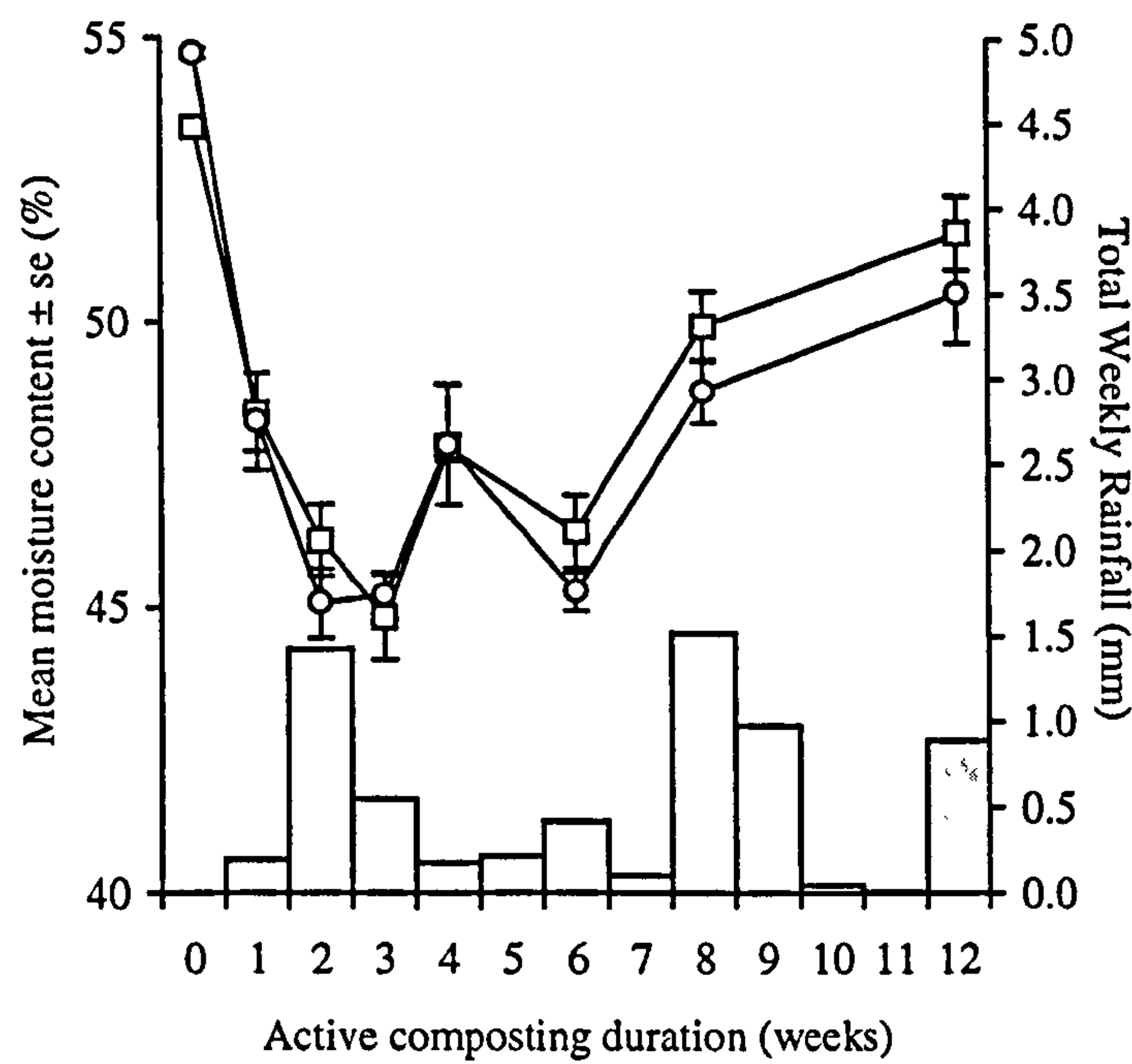
Figure 2 – Changes in moisture content (treated windrow –○–; control windrow –□–)



Mean moisture contents of the treated windrow and control were initially 68 and 72% respectively. The mean moisture content of the treated windrow fell to 48% at week 6; so water was added to prevent moisture falling to inhibitory levels (section 2.2.3), raising the mean moisture content to 55%. The mean moisture content of the control remained between 70–75% during the 12 week experimental period. The mean moisture content of the treated windrow was significantly below that recorded for the control at all sampling periods ($p < 0.05$), with a maximum difference of 22.5%.

Experiment 3

Figure 3 – Changes in moisture content (treated windrow –O–; control windrow –□–)

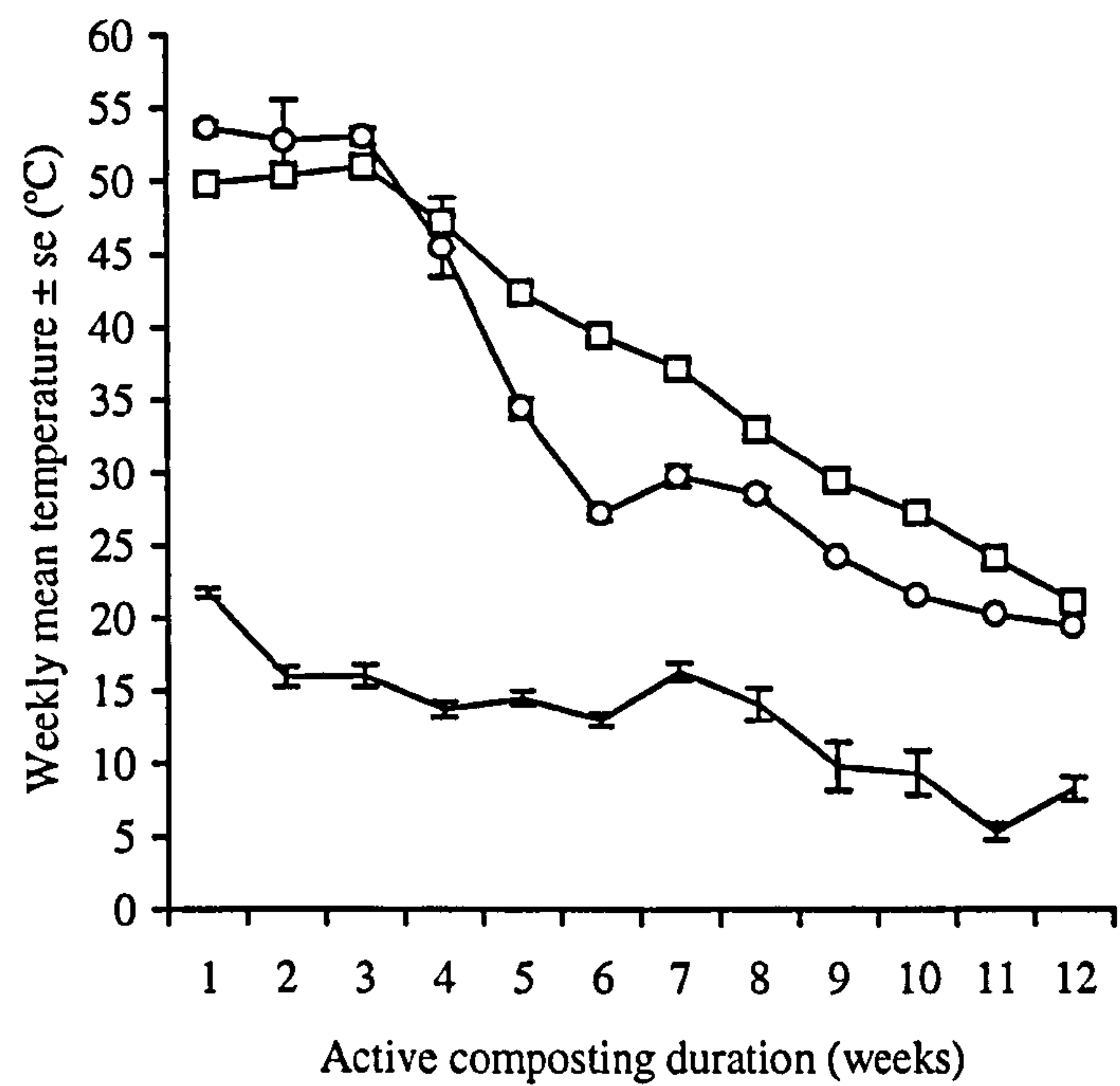


Mean moisture contents of the treated windrow and control were initially 55 and 53% respectively. During the 12 week composting period mean moisture contents remained between 45–55% (treated windrow) and 45–53% (control). The mean moisture contents of the treated windrow and control were not significantly different at any sampling period ($p > 0.05$).

2.3.2 Temperature

Experiment 1

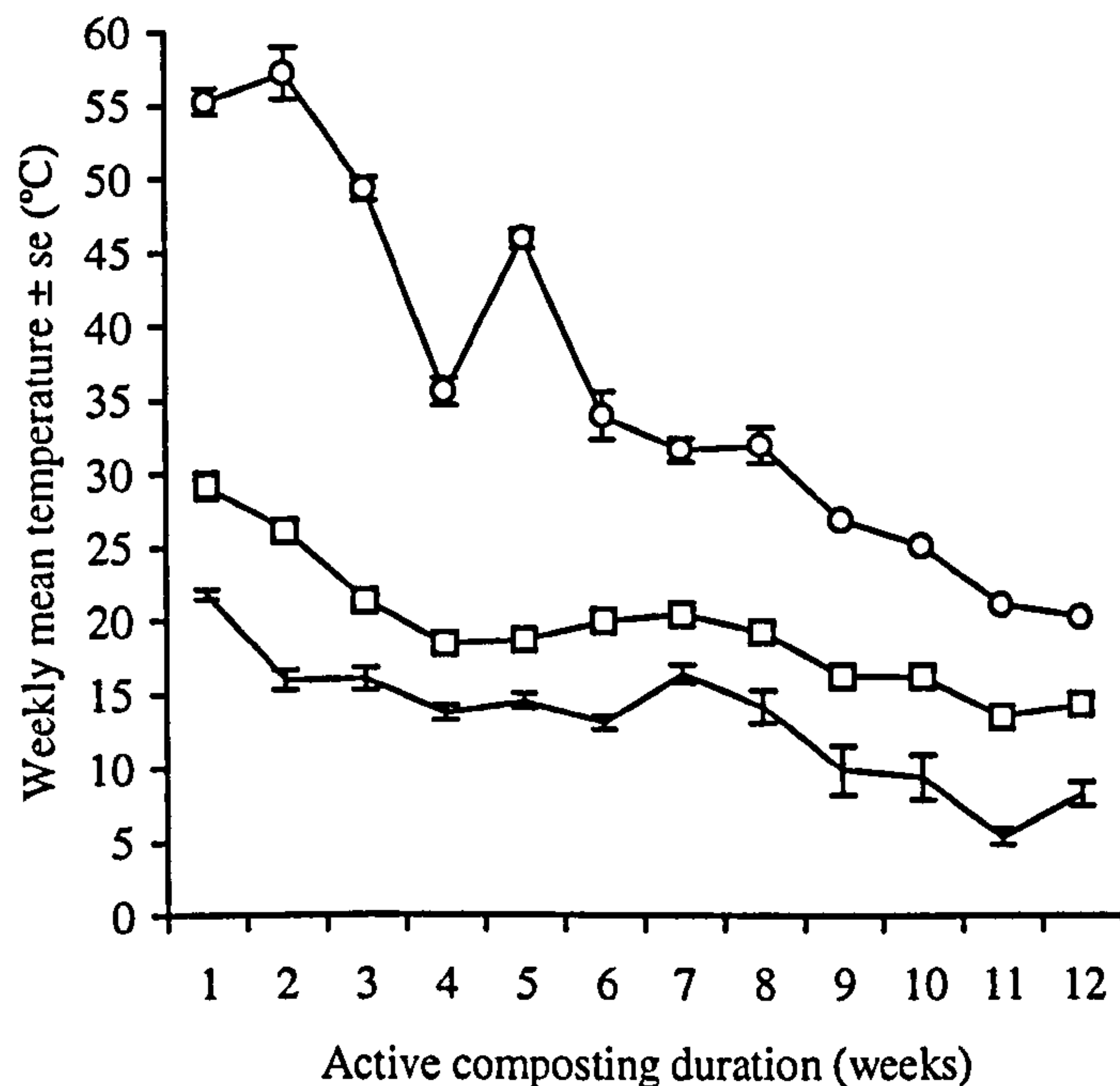
Figure 4 – Changes in windrow temperature (treated windrow –○–; control windrow –□–; ambient temperature —)



Mean temperatures of between 46–54°C (treated windrow) and 47–51 (control) were recorded over the first 4 weeks of composting; with no significant difference ($p > 0.05$). However, after week 4 temperatures in the treated windrow and control diverged. The treated windrow temperature decreased more rapidly than the control, and by week 6 the temperature of the treated windrow had fallen 12°C below that of the control ($p < 0.05$). At week 8 temperatures were no longer significantly different between windrows ($P > 0.05$). During the 4 week maturation period (weeks 9–12) the mean temperature in both the treated windrow and control remained below 30°C.

Experiment 2

Figure 5 – Changes in windrow temperature (treated windrow –○–; control windrow –□–; ambient temperature —)

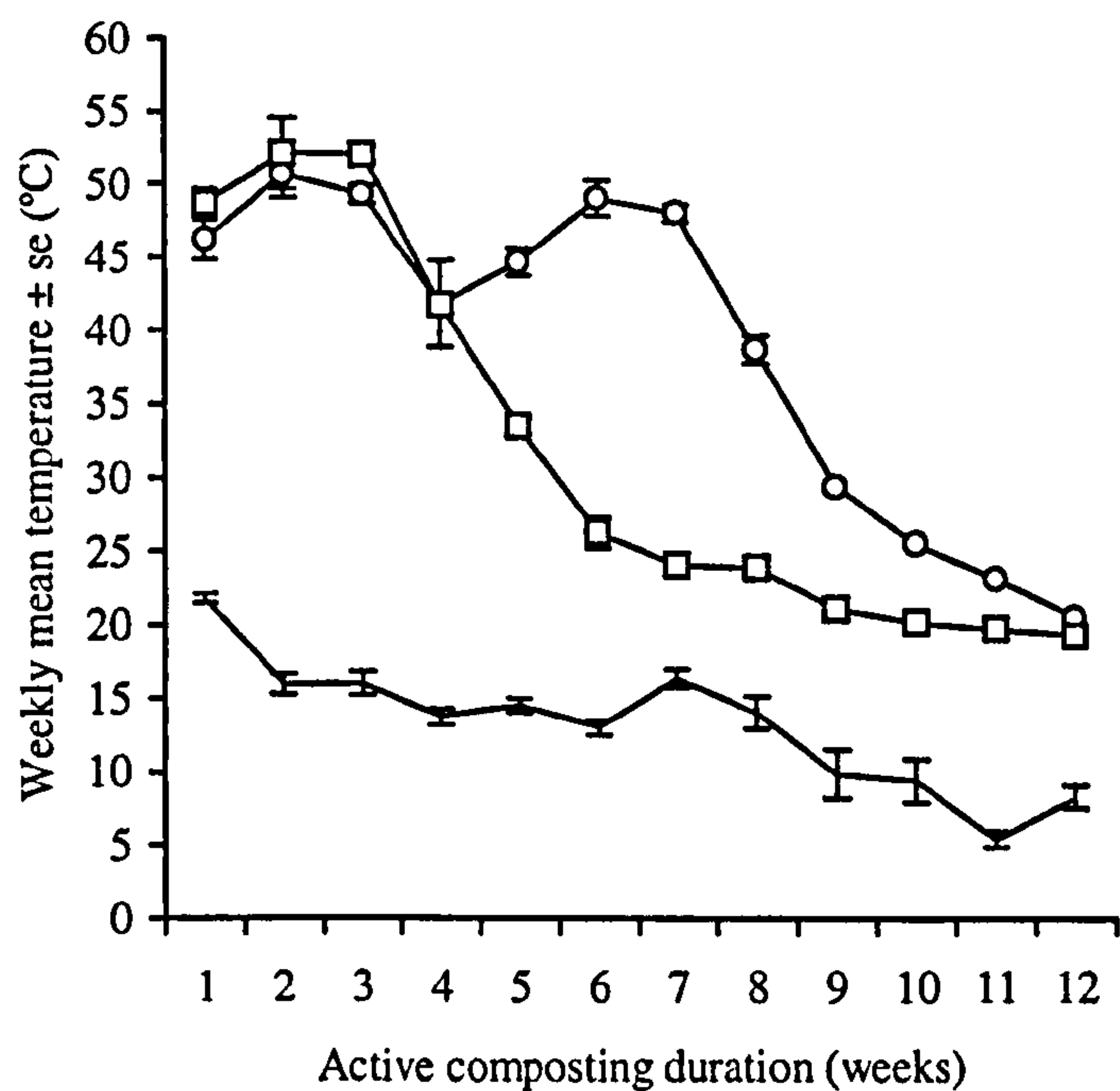


Mean temperatures in the treated windrow of 49–57°C were recorded over the first 3 weeks of active composting. The treated windrow temperature fell to 36°C by week 4 with a large variation in temperature readings (95% confidence intervals = 27 & 45°C). After the addition of water at week 4 (section 2.2.3) treated windrow temperature rose to 46°C by week 5, and then fell gradually for the remaining experimental period, reaching a minimum of 20°C at week 12.

Mean temperatures in the control, falling within a range of 10–30°C, were significantly lower than the treated windrow at all sampling periods ($p < 0.05$). Linear regression analysis over the whole 12 week composting period suggested that control temperatures were highly dependant upon ambient temperature, following the equation: control temp. = $0.96 \times$ ambient temp. + 6.82°C ($R^2 = 0.85$, $S_{(x,y)} = 1.86^\circ\text{C}$; slope differs from zero significantly $p < 0.0001$; y intercept differs from zero significantly $p < 0.01$).

Experiment 3

Figure 6 – Changes in windrow temperature (treated windrow –○–; control windrow –□–; ambient temperature —)



Mean temperatures of between 42–51°C were recorded in the treated windrow for the first 7 weeks of active composting, falling thereafter to between 20–30°C during the maturation period (week 9–12).

Mean temperatures between 42–52°C were recorded in the control over the first 4 weeks of active composting, with no significant difference from the treated windrow. The control mean temperature then decreases to between 20–30°C from week 6 onwards, whereas the treated windrow mean temperature increased before dropping from week 8 through to week 12.

Control windrow mean temperatures were significantly below those of the treated windrow from week 5 to 8 inclusive, with a maximum difference of 24°C at week 7 ($p < 0.01$). Treated windrow and control mean temperatures converge at the end of the composting period (week 12) to 19 and 20°C respectively, with no significant difference between them ($p > 0.05$).

2.3.3 Total organic matter

Table 3 – Loss of total organic matter (%ash*).

Expt. 1			Expt. 2		Expt. 3	
Week	Treated	Control	Treated	Control	Treated	Control
0.5	6.0±0.4	2.1±0.8	8.4±2.2	2.0±1.0	5.6±0.7	5.5±0.8
1.0	10.9±0.5	4.9±0.5	19.2±1.9	5.3±0.1	8.1±0.7	11.4±0.6
1.5	15.6±0.9	7.5±1.1	28.8±2.6	8.5±0.9	10.5±0.8	16.9±0.8
2.0	19.9±1.0	8.9±0.7	29.7±1.0	16.3±0.8	12.8±0.3	21.6±0.4
3.0	23.3±0.9	11.7±1.0	34.1±0.7	18.0±0.9	12.3±0.7	24.4±0.4
4.0	25.0±0.8	18.6±1.0	41.6±0.6	18.5±1.4	19.2±0.8	26.4±0.3
6.0	23.3±0.1	19.7±0.3	43.4±0.7	20.3±1.3	21.4±1.1	25.8±1.0
8.0	28.5±0.9	21.8±0.2	47.1±1.7	19.1±1.0	27.3±0.8	28.4±0.5
12.0	30.0±0.6	25.3±0.5	48.8±0.2	22.9±1.0	30.7±0.5	30.5±0.3

* calculations based on equations in section 2.2.4.0.0.

Table 4 – Non-linear regression, TOM (%DM) vs. composting duration.

Variable	Expt. 1		Expt. 2		Expt. 3	
	Treated	Control	Treated	Control	Treated	Control
Total TOM loss (a)	7.96±0.30	7.65±0.35	15.74±0.51****	6.35±0.36	9.89±0.72	8.32±0.23
Final TOM (c)	34.37±0.22	34.37±0.38	32.21±0.41****	36.38±0.26	31.69±0.80*	33.99±0.15
Decay rate (λ)	0.49±0.05****	0.21±0.02	0.41±0.03	0.46±0.07	0.14±0.02****	0.55±0.04
Half-life (weeks)	1.41±0.14	3.29±0.31	1.68±0.12	1.50±0.23	4.93±0.70	1.25±0.09
Measures of goodness of fit						
R²	0.95	0.96	0.96	0.90	0.96	0.97
S _{x,y} (TOM%DM)	0.63	0.47	1.05	0.74	0.54	0.48
Runs test	p = 0.44	p = 0.16	p = 0.32	p = 0.14	p < 0.05	p < 0.01

* = p < 0.05, **** = p < 0.0001; t-test comparing treated and control windrows within individual experiments.

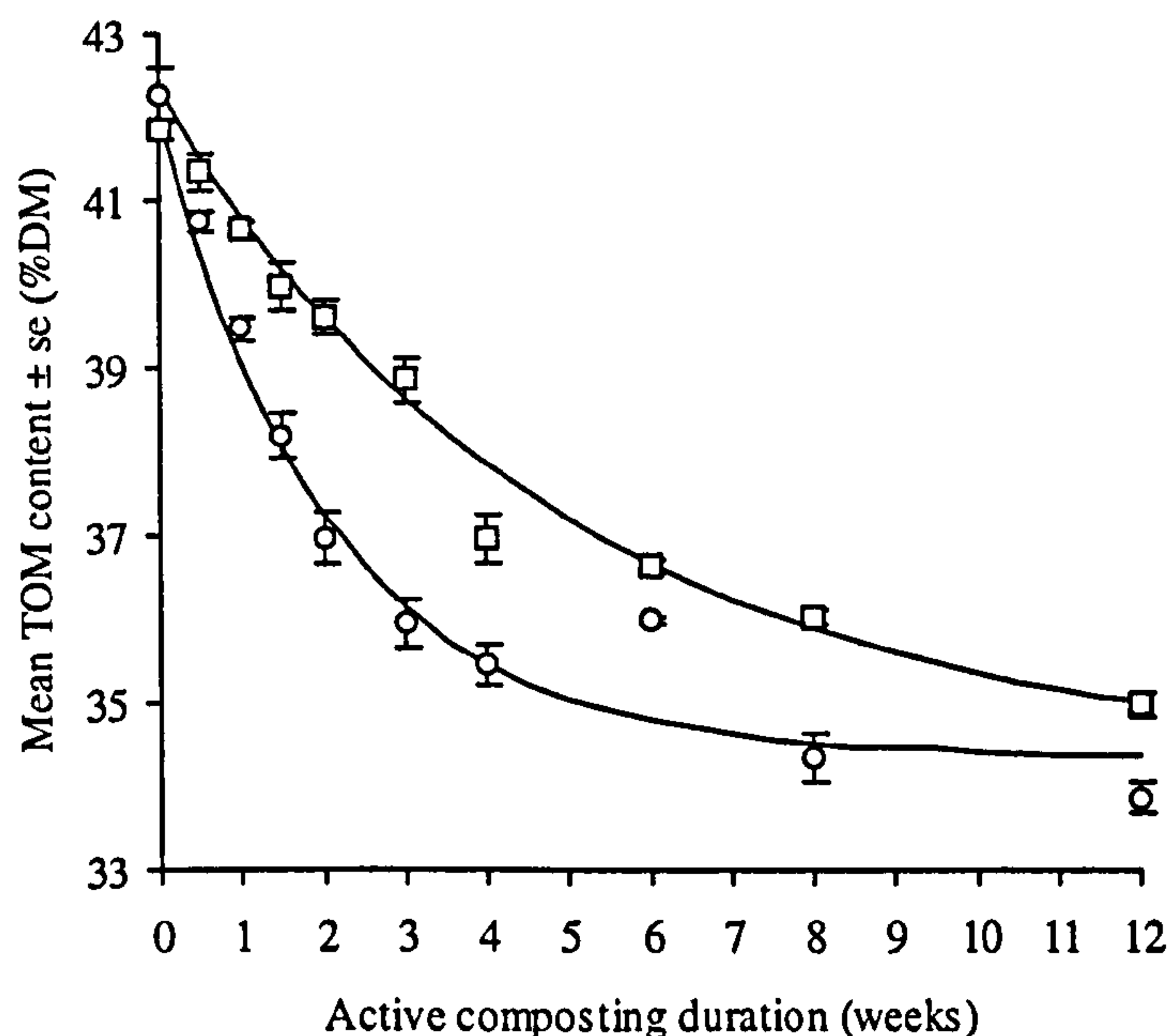
Runs test = probability that the experimental data differs significantly from the model (p < 0.05).

R² = fraction of total variance in experimental data explained by the model.

S_{x,y} = standard deviation of points from the model line-of-best-fit. (Motulsky 1996)

Experiment 1

Figure 7 – Changes in TOM (%DM) with composting duration, (treated windrow O; control windrow □; one phase decay model line of best fit —).

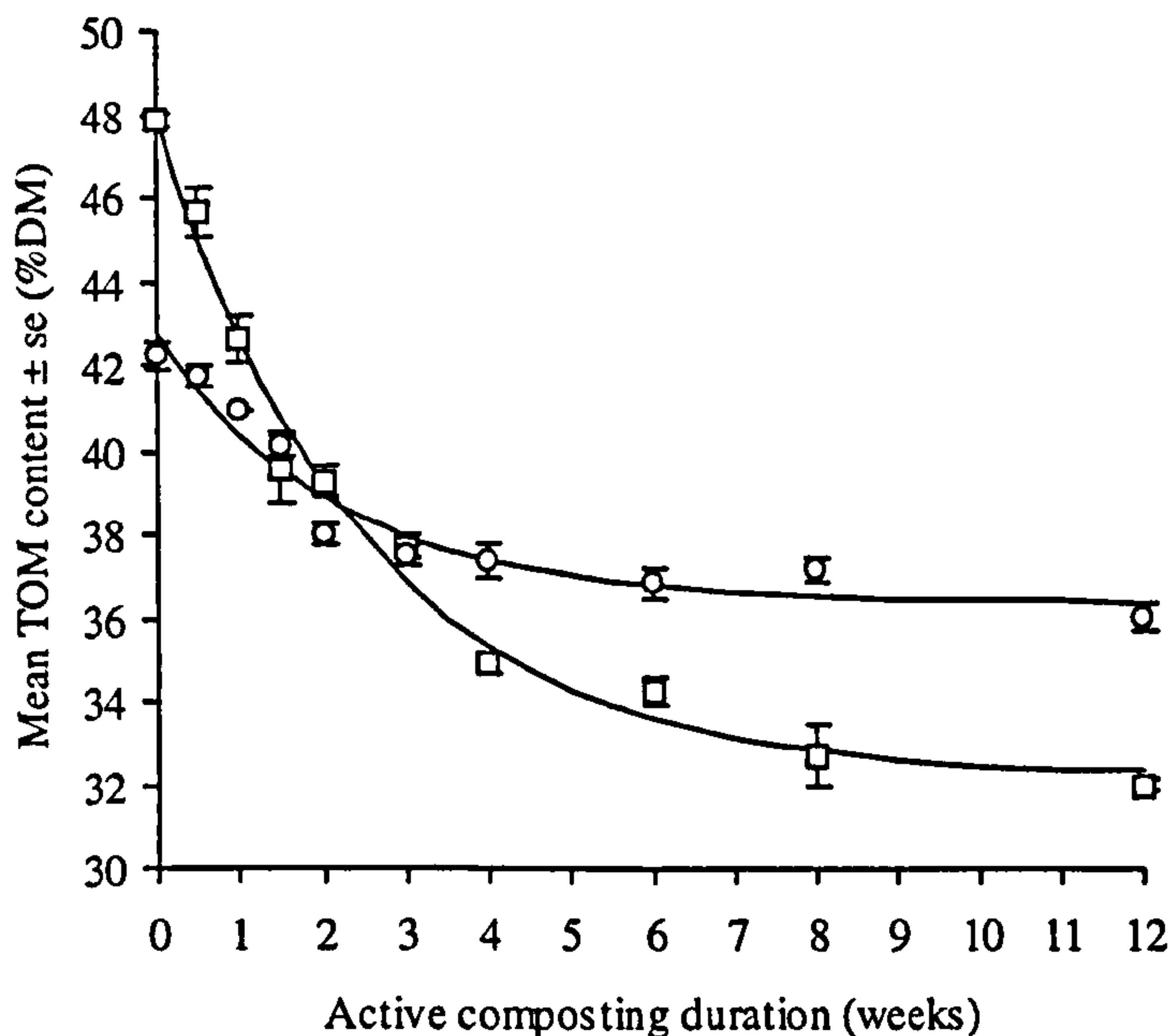


Results of non-linear regression show that the one-phase-exponential decay model provides a good fit for the TOM (%DM) data (Table 4). The model suggested that total TOM%DM losses and final TOM levels were not significantly different between the treated windrow and the control. However, the treated windrow showed a significantly greater rate of decay ($p < 0.0001$, $t_{(14df)} = 5.39$) with a 2.3 fold mean difference in half-life.

Individual t-tests show initial TOM (%DM) contents were not significantly different. Samples taken after 8 weeks composting were significantly different ($p < 0.01$) but not after a further 4 weeks maturation. Loss of TOM (%ash) supports the non-linear model, with greater losses shown within the treated windrow in the first 4 weeks of composting, 25.0%ash (6.8%DM) in the treated windrow and 18.6%ash (4.9%DM) in the control (Table 3). A further mean reduction of only 5.0%ash (1.5%DM) in the treated windrow and 6.7%ash (2.0%DM) in the control was recorded for the remainder of the 12 week period.

Experiment 2

Figure 8 – Changes in TOM (%DM) with composting duration; (treated windrow ○; control windrow □; one phase decay model line of best fit —).



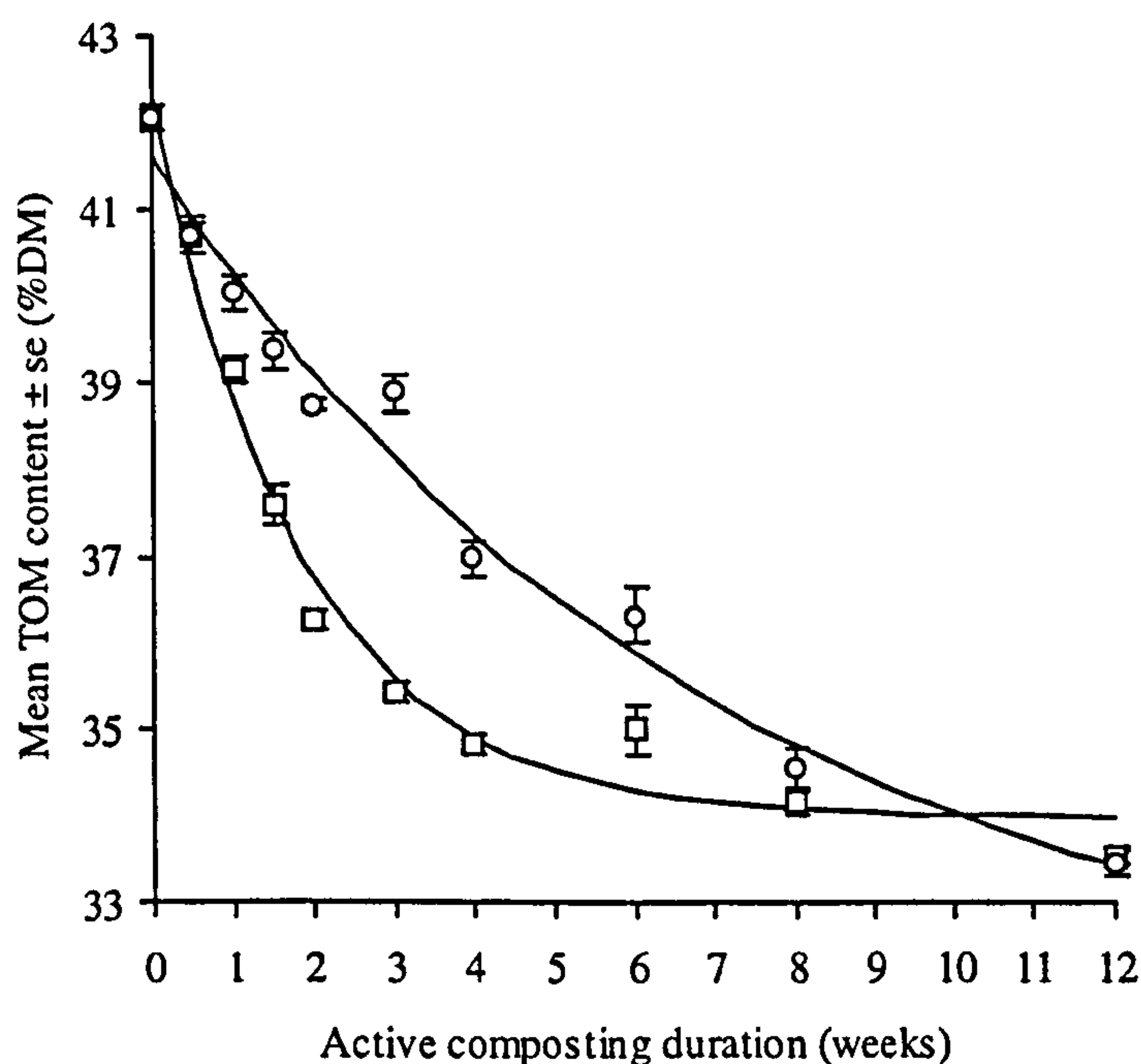
The non-linear regression results show that the one-phase-exponential decay model provides a good fit for the TOM (%DM) data (Table 4). The model suggested that total TOM%DM losses and final TOM levels were very significantly different between the treated windrow and the control ($p < 0.0001$, $t_{(14df)} = 14.99$). However, the treated windrow showed no significantly greater rate of decay than the control, with a similar mean half-life. This suggests the time taken for the total loss of TOM in the treated and control windrows was similar.

Individual t-tests show initial TOM (%DM) contents were significantly different ($p < 0.0001$), with a mean difference of 5.6%DM. The higher TOM%DM value of the treated windrow was due to the addition of barley-straw, which had a much higher TOM content than 2°WPS (section 2.2.1). At week 1.5, despite the treated windrow's higher starting value, TOM%DM contents were not statistically different ($p > 0.05$). Samples taken after 12 weeks composting were statistically different ($p < 0.0001$), with a mean difference of 4.1%DM.

Loss of TOM (%ash) measurements shows greater losses in the treated windrow during the first 4 weeks of composting: 41.6%ash (13.0%DM) in the treated windrow and 18.5%ash (4.9%DM) in the control (Table 3). A further mean reduction of only 7.2%ash (2.9%DM) in the treated windrow and 4.4%ash (1.3%DM) in the control was recorded for the remainder of the 12 week period.

Experiment 3

Figure 9 – Changes in TOM (%DM) with composting duration; treated windrow ○; control windrow □; one phase decay model line of best fit —.



Results of non-linear regression show that the one-phase exponential-decay model does not strictly provide a true fit for the TOM (%DM) data (Runs test, $p < 0.05$). However, as other regression models used (logistic and linear) also did not provide true fits, the same model was used. The model suggested that total TOM losses and final TOM levels were not significantly different between the treated windrow and control. However, the control windrow appeared to show a significantly greater rate of decay ($p < 0.0001$, $t_{(14df)} = 9.41$) with a 3.9-fold difference in mean half-life.

Individual t-tests show initial and final (12 weeks) TOM (%DM) contents were not significantly different.

Loss of TOM (%ash) conforms to the non-linear model, with greater losses within the control windrow in the first 4 weeks of composting, 26.4%ash (7.2%DM) in the control and 19.2%ash (5.0%DM) in the treated windrow (Table 3). A further mean reduction of only 4.1%ash (1.3%DM) in the control and 11.5%ash (3.5%DM) in the treated windrow was recorded for the remainder of the 12 week period.

2.3.4 Nitrogen content

Table 5 – Changes in nitrogen content, in varying forms*; mean±se.

Windrow	Sample (week)	TKN %	TOC %	C/N ratio	Water soluble		
					NH ₄ ⁺ mg/kg	NO ₃ ⁻ mg/kg	NH ₄ ⁺ /NO ₃ ⁻ ratio
Expt. 1	0	0.64±0.05	23.5±0.4	37±3	3829±341	7±0	526
Treated	4	0.41±0.01	19.7±0.3	48±0	1229±136	9±1	139
	8	0.41±0.01	19.1±0.3	47±1	28±7	253±37	0.11
Control	0	0.35±0.05	23.3±0.2	67±1	22±4	7±1	3
	4	0.43±0.01	20.6±0.3	48±2	21±4	8±0	3
	8	0.38±0.02	20.0±0.1	53±3	41±5	10±0	4
Expt. 2	0	1.68±0.10	26.6±0.2	16±1	5317±206	9±1	571
Treated	4	1.45±0.04	19.4±0.27	13±0	1878±50	10±2	181
	8	1.22±0.10	18.2±0.8	15±2	160±14	290±45	0.55
Control	0	2.62±0.08	23.5±0.3	9±0	6110±199	9±1	705
	4	2.70±0.13	20.7±0.4	8±0	6511±192	9±2	689
	8	2.25±0.06	20.7±0.3	9±0	4780±670	8±0	575
Expt. 3	0	0.78±0.07	23.3±0.1	30±2	4831±111	5±0	919
Treated	4	0.68±0.02	20.6±0.3	30±1	2715±145	7±0	370
	8	0.62±0.03	19.2±0.3	31±2	579±105	153±19	4
Control	0	0.69±0.04	23.3±0.1	34±2	1160±37	7±0	178
	4	0.69±0.01	19.3±0.3	28±1	1067±39	9±1	112
	8	0.58±0.03	19.0±0.2	33±2	84±13	145±8	0.58

*All figures expressed in terms of dry mass where applicable.

Experiment 1

Initial levels of TKN (Table 5) were 0.29%DM greater in the treated windrow ($p < 0.001$), with an initial C/N ratio almost half (56%) of that recorded for the control. The difference in TKN was accounted for by an increase of 3807 mg kg⁻¹DM in ammonium, equating to a total nitrogen content of 0.30%DM. The increase in ammonium was largely due to the breakdown of urea CO(NH₂)₂ added as part of the fertiliser treatment (section 2.2.1).

After 4 weeks of active composting the treated windrow had lost 0.23%DM (43%ash of its initial level) TKN ($p < 0.001$); accounted for by a 2600 mg kg⁻¹ (71%ash of its initial level) net reduction in NH₄⁺ ($p < 0.001$), equivalent to a loss of 0.20%DM total nitrogen. The loss in TKN, approximately equal to the loss of nitrogen in the form of NH₄⁺, suggests the net decrease in NH₄⁺ was via volatilisation or leaching.

After 8 weeks active composting the treated windrow showed a further 1201 mg kg⁻¹ decrease in NH₄⁺ ($p < 0.05$) with no significant difference in TKN ($p > 0.05$). This suggests net NH₄⁺

loss was via immobilisation and nitrification. The onset of nitrification was indicated by a significant increase in NO_3^- ($p < 0.05$).

Nitrogen content of the control windrow did not differ significantly from initial values ($p > 0.05$). Although there was a significant increase in NH_4^+ ($p < 0.05$), this was small (19 mg kg^{-1}). Nitrification did not occur in the control windrow.

There was a significant reduction in C/N ratio in the control windrow ($p < 0.05$) over the 8 week active composting period, indicating that the relative loss in carbon exceeded total nitrogen loss. The C/N ratio of the treated windrow increased over the 8 week active composting period, indicating that the loss of nitrogen exceeded the relative loss of carbon. However, the C/N ratio of the treated windrow was significantly lower than the control ($p < 0.01$) after 8 weeks composting.

Experiment 2

The TOC content of the treated windrow was lower than the 27.3%DM TOC estimated during mixture formulation (section 2.2.1), this was probably due to errors in physicochemical determinations of feedstock materials. However, the initial TKN content (1.68%DM) was much lower than estimated (2.28%DM), giving a C/N ratio (15.8) higher than estimated (12). This suggests a Nitrogen loss may have occurred during mixture preparation and windrow formation.

Initial levels of TKN were 0.94%DM lower in the treated windrow ($P < 0.001$), giving an initial C/N ratio 1.8 fold higher than the control.

After 4 weeks of active composting, the treated windrow had lost 0.52%DM TKN, (31%ash of initial level) ($p < 0.001$). This was largely accounted for by a 3439 mg kg^{-1} (72%ash of initial level) net reduction in NH_4^+ ($p < 0.001$), equivalent to a loss of 0.30%ash total nitrogen. The loss in TKN exceeds the loss of nitrogen in the form of NH_4^+ , which suggests net NH_4^+ losses were via volatilisation or leaching. Despite a large loss in nitrogen the C/N ratio decreased, indicating a relatively greater loss of TOC had occurred.

There were no significant changes in TKN or NH_4^+ content within the control windrow after 4 weeks of active composting ($p > 0.05$). A decrease in C/N ratio was observed, due to a significant loss of TOC (section 2.3.3).

After 8 weeks of active composting, the treated windrow showed a further decrease of 0.23%DM TKN, (13%ash of initial level) ($p < 0.05$), accounted for by a 1718 mg kg^{-1} net loss in NH_4^+ (26%ash of initial level), equivalent to a net loss of 0.11%ash in total nitrogen. The disparity in total nitrogen lost in the form of TKN and NH_4^+ suggest NH_4^+ was formed from the breakdown of organic matter. The onset of nitrification was indicated by a 280 mg kg^{-1}

increase in NO_3^- . The rise in C/N ratio from week 4 suggests nitrogen loss exceeded TOC loss during this period. The final C/N ratio was not significantly different from the starting C/N ratio ($p > 0.05$).

After 8 weeks of active composting, the control windrow showed a decrease of 0.55%DM TKN (21%ash of initial level) ($p < 0.05$), accounted for by a 1731 mg kg^{-1} (28%ash of initial level) net loss in NH_4^+ , equivalent to a net loss of 0.13%DM total nitrogen. The disparity in total nitrogen lost in the forms of TKN and NH_4^+ suggests NH_4^+ was formed from the breakdown of organic matter. There was no significant increase in NO_3^- ($p > 0.05$). The final C/N ratio was not significantly different from the starting C/N ratio ($p > 0.05$), indicating nitrogen loss was matched by TOC loss.

Experiment 3

Initial TKN contents were lower than estimated for both control and treated windrows, giving C/N ratios higher than estimated (section 2.2.1). This suggests a loss of nitrogen during mixture preparation and windrow formation. There was no statistical difference between the control and treated windrows TKN or C/N ratios.

There was, however, a 3671 mg kg^{-1} higher level of NH_4^+ recorded for the treated windrow, equivalent to 0.29%DM total nitrogen. The difference in levels of NH_4^+ appeared to be only partially reflected in the relative levels of TKN recorded. After 4 weeks of active composting, the control windrow showed no significant change in TKN or NH_4^+ . Although a significant reduction in C/N indicated a significant decrease in TOC (section 2.3.3.c.)

After 8 weeks of active composting, the treated windrow showed a decrease of 0.18%DM TKN ($p < 0.05$), partially accounted for by an overall loss of 1076 mg kg^{-1} (94%ash of initial level) NH_4^+ , equivalent to a net loss of 0.08%DM total nitrogen. The disparity in total nitrogen lost in the form of TKN and NH_4^+ suggest NH_4^+ was formed from the breakdown of organic matter. The onset of nitrification was indicated by a 138 mg kg^{-1} increase in NO_3^- . The rise in C/N ratio from week 4 suggests nitrogen loss exceeded TOC loss during this period. The final C/N ratio was not significantly different from the starting C/N ratio ($p > 0.05$).

2.3.5 Macronutrient content (P, K, Mg)

Table 6 – Phosphorus (P), potassium (K) and magnesium (Mg) contents

Windrow	Sample (week)	Total		Water soluble			Nitrogen/Nutrient ratios	
		P %	K %	P mg kg ⁻¹	K mg kg ⁻¹	Mg mg kg ⁻¹	N/P	N/K
Expt. 1 Treated	0	0.33±0.02	0.46±0.01	548±9	2441±65	15±1	1.94±0.03	1.39±0.08
	4	0.29±0.02	0.40±0.02	56±4	1409±76	22±7	1.46±0.06	1.03±0.05
	8	0.31±0.01	0.41±0.05	8±0	1208±67	59±5	1.32±0.01	1.00±0.16
Control	0	0.05±0.01	0.14±0.01	0±0	33±2	29±1	6.80±0.30	2.46±0.00
	4	0.07±0.02	0.16±0.01	6±0	51±2	52±2	6.14±1.25	2.64±0.09
	8	0.07±0.01	0.18±0.01	9±0	48±2	26±1	5.43±0.43	2.13±0.11
Expt. 2 Treated	0	1.00±0.07	0.70±0.01	379±67	2836±98	55±1	1.67±0.02	2.38±0.07
	4	1.05±0.06	0.81±0.01	151±8	2399±37	54±3	1.37±0.04	1.77±0.03
	8	1.09±0.03	0.76±0.03	27±3	2255±202	145±20	1.15±0.06	1.66±0.00
Control	0	1.33±0.12	0.35±0.01	231±23	859±21	79±7	1.98±0.11	7.59±0.01
	4	1.40±0.00	0.40±0.01	602±13	1034±36	60±11	1.93±0.09	6.84±0.02
	8	1.35±0.05	0.38±0.01	296±71	992±35	55±9	1.50±0.01	5.35±0.02
Expt. 3 Treated	0	0.55±0.07	0.52±0.02	500±27	2244±83	19±2	1.42±0.05	1.50±0.05
	4	0.58±0.04	0.58±0.00	489±12	2171±44	29±4	1.17±0.04	1.17±0.01
	8	0.60±0.05	0.59±0.06	52±2	1935±181	81±10	1.03±0.03	1.05±0.12
Control	0	0.26±0.03	0.18±0.00	40±3	150±8	33±5	2.65±0.14	3.83±0.01
	4	0.33±0.03	0.21±0.01	135±6	187±4	18±1	2.09±0.15	3.29±0.10
	8	0.34±0.02	0.24±0.01	17±1	117±6	69±2	1.71±0.01	2.42±0.02

Experiment 1

Initially, total phosphorus (P) and potassium (K) contents in the treated windrow were similar to those estimated during treatment formulation (section 2.2.1). The addition of fertiliser increased levels of total P and K in the treated windrow with correspondingly lowered N/P and N/K ratios. The greater level of total potassium (0.32%DM) in the treated windrow was largely present in water soluble form (2408 mg kg⁻¹ = 0.24%DM), whereas only a small fraction of the greater level of total phosphorus (0.28%DM) was present in water soluble form (548 mg kg⁻¹ = 0.055%DM).

During 8 weeks active composting there was a significant (p < 0.001) reduction in water soluble P in the treated windrow, corrected for DM loss as 99%ash (543 mg kg⁻¹). Although there was no significant loss in total P (%DM), correcting for DM loss suggested a 19%ash (627 mg kg⁻¹) mean loss in total P. Water soluble K significantly decreased within the treated windrow (p < 0.001), corrected for DM loss as 57%ash (1391 mg kg⁻¹) after 8 weeks active composting. Changes in total potassium (%DM) were not significant, but correcting for DM loss suggested a 24%ash (1104 mg kg⁻¹) mean loss in total K. The low level of water soluble magnesium (Mg) increased 4-fold within the treated windrow (p<0.05).

Within the control windrow there were no significant changes in P, K, or Mg nutrient levels ($p > 0.05$).

Experiment 2

Initially total phosphorus (P) and potassium (K) contents in the treated windrow were similar to those estimated during treatment formulation (section 2.2.1). The addition of straw resulted in increased levels of total K and a reduction in total P within the treated windrow. Of the greater level of total potassium (0.35%DM) in the treated windrow, 56% was present in water soluble form ($1977 \text{ mg kg}^{-1} = 0.20\% \text{DM}$). Although there was a significantly lower level of total P (0.33%DM) within the treated windrow ($p < 0.05$), there was a 148 mg kg^{-1} (0.015%DM) greater level in mean water soluble P ($p < 0.05$).

After 8 weeks active composting there was a significant reduction in water soluble P in the treated windrow ($p < 0.001$), corrected for DM loss as 94%ash (356 mg kg^{-1}). Although there was no significant change in total P (%DM), correcting for DM loss suggested a 17%ash (1700 mg kg^{-1}) mean loss in total P, far exceeding the net loss of water soluble P. This suggested high levels of P mineralisation during decomposition, and a loss of P through leaching. Water soluble K (mg kg^{-1}) significantly decreased over 8 weeks active composting within the treated windrow ($p < 0.001$), corrected for DM loss as 43%ash (1219 mg kg^{-1}). Although there was no significant change in total K (%DM), correcting for DM loss suggested a 18%ash (1260 mg kg^{-1}) mean loss. A significant increase in water soluble Mg was observed in the treated windrow over 8 weeks active composting.

Within the control windrow there was little change in P, K, or Mg over 8 weeks active composting. However, there were significant increases in water soluble P and K after 4 weeks ($p < 0.05$), which decreased by week 8 to levels broadly similar to those recorded for fresh WPS.

Experiment 3

Initially total P and K contents in the treated windrow were similar to those estimated during treatment formulation (section 2.2.1). The addition of fertiliser increased levels of total P and K in the treated windrow. The greater level of total K (0.34%DM) in the treated windrow was largely present in water soluble form (0.21%DM). The treated windrow also contained a significantly ($p < 0.05$) higher level of total phosphorus (0.29%DM), only a small fraction of which was present in water soluble form (0.046%DM).

During 8 weeks active composting there was a significant ($p < 0.001$) reduction in water soluble P in the treated windrow, corrected for DM loss as 91%ash (455 mg kg^{-1}); there was no significant change in total P. Although there was no significant change in water soluble K

(mg kg⁻¹), correcting for DM loss suggested a loss of 24%ash (586 mg kg⁻¹) over 8 weeks active composting; no significant loss in total K was observed. The level of water soluble magnesium (Mg) increased significantly within the treated windrow ($p < 0.05$).

Within the control windrow there were significant ($p < 0.05$), though small, decreases in water soluble P and K over 8 weeks composting. There were no significant changes in total P and K. Water soluble Mg increased significantly over 8 weeks active composting.

2.3.6 Changes in acidity/alkalinity (pH) and electrical conductivity (EC).

Table 7. – Changes in pH during active composting

Windrow	Sample (week)	pH	H ⁺
Expt. 1 Treated	0	8.9±0.0	1.3 × 10 ⁻⁹
	4	8.7±0.1	1.9 × 10 ⁻⁹
	8	7.9±0.0	1.3 × 10 ⁻⁸
Control	0	7.9±0.0	1.3 × 10 ⁻⁸
	4	7.6±0.0	2.6 × 10 ⁻⁸
	8	7.9±0.0	1.3 × 10 ⁻⁸
Expt. 2 Treated	0	8.7±0.0	2.1 × 10 ⁻⁹
	4	8.3±0.1	5.2 × 10 ⁻⁹
	8	7.7±0.0	2.3 × 10 ⁻⁸
Control	0	8.0±0.1	9.9 × 10 ⁻⁹
	4	8.1±0.1	7.6 × 10 ⁻⁹
	8	8.1±0.0	8.1 × 10 ⁻⁹
Expt. 3 Treated	0	8.8±0.0	1.6 × 10 ⁻⁹
	4	8.7±0.0	1.9 × 10 ⁻⁹
	8	7.6±0.1	2.8 × 10 ⁻⁸
Control	0	8.5±0.1	3.4 × 10 ⁻⁹
	4	8.7±0.0	1.9 × 10 ⁻⁹
	8	7.5±0.0	3.4 × 10 ⁻⁸

Experiment 1

Initially, pH was one unit higher in the experimental windrow, a ten-fold difference in [H⁺]. This was caused by high levels of ammonium present in the treated windrow (section 2.3.4). During 8 weeks active composting the pH decreased ([H⁺] increased) significantly within the treated windrow (p < 0.0001), to the same level as the original and final pH within the unamended control windrow. The greatest reduction in pH came between weeks 4 and 8 from a value of 8.7 to a value of 7.9.

A significant reduction in pH occurred within the control windrow after 4 weeks active composting, equating to a two-fold increase in [H⁺]. However, after 8 weeks composting pH had returned to its initial level.

Final pH levels of the treated and control windrows were not significantly different.

Experiment 2

Initially pH was higher in the experimental windrow, with a 4.7-fold difference in $[H^+]$, possibly due to higher levels of ammonium present in the treated windrow (section 2.3.4).

During 8 weeks active composting the pH level decreased significantly within the treated windrow ($p < 0.0001$), equating to a ten-fold increase in H^+ ions. The greatest reduction in pH came between weeks 4 and 8 from 8.3 to 7.7.

No significant change in pH occurred within the control windrow during active composting.

The final pH level of the control windrow was significantly higher than the treated windrow ($p < 0.001$).

Experiment 3

Initially pH was significantly higher in the experimental windrow, with a 2.1-fold difference in $[H^+]$, possibly due to higher levels of ammonium present in the treated windrow (section 2.3.4).

During 8 weeks active composting the pH level decreased significantly within the treated windrow ($p < 0.001$), equating to a 17.5-fold increase in H^+ ions. The greatest reduction in pH came between weeks 4 and 8 from 8.7 to 7.6.

Significant reduction in pH occurred within the control windrow during 8 weeks active composting ($p < 0.0001$), equating to a 10-fold increase in H^+ .

The final pH levels of the control and treated windrows were not significantly different.

Table 8 – Changes in electrical conductivity (EC) during composting

Windrow	Sampling (week)	EC $\mu\text{S cm}^{-1}$
Expt. 1	0	2088 \pm 24
Treated	4	1055 \pm 87
	8	583 \pm 24
Control	0	191 \pm 2
	4	295 \pm 5
	8	211 \pm 7
Expt. 2	0	2363 \pm 52
Treated	4	1648 \pm 31
	8	1108 \pm 124
Control	0	2588 \pm 52
	4	3038 \pm 85
	8	2013 \pm 157
Expt. 3	0	3150 \pm 84
Treated	4	1988 \pm 75
	8	1160 \pm 134
Control	0	734 \pm 12
	4	828 \pm 12
	8	546 \pm 25

Experiment 1

Initial EC was 10.9 fold higher in the experimental windrow ($p < 0.05$), due to higher levels of water soluble nutrients added.

A Pearson’s rank correlation analysis of the total concentration of all water soluble nutrients ($\text{NH}_4^+ + \text{NO}_3^- + \text{P}^{3-} + \text{K}^+ + \text{Mg}^{2+}$) with EC, recorded for all samples from all experiments, showed a significant positive correlation (Pearson $r = 0.97$; $p < 0.0001$; $R^2 = 0.93$).

During 8 weeks active composting the EC level decreased by 1505 $\mu\text{S cm}^{-1}$ (72%) within the treated windrow ($p < 0.05$). This reflects an overall reduction of water soluble nutrients during composting (*q.v.* sections 2.3.4.0 & 2.3.5.0).

There was no significant change from initial EC within the control windrow after 8 weeks active composting.

The final EC of the treated windrow was significantly higher than the control windrow ($p < 0.05$).

Experiment 2

Table 8 shows a small (225 $\mu\text{S cm}^{-1}$), though statistically higher level of EC in the control windrow ($p < 0.05$).

During 8 weeks active composting the EC level decreased by $1255 \mu\text{S cm}^{-1}$ (53%) within the treated windrow ($p < 0.05$). This reflects an overall reduction in water soluble nutrients during composting (*q.v.* sections 2.3.4.0 & 2.3.5.0).

A smaller decrease of $575 \mu\text{S cm}^{-1}$ (22%) from initial EC occurred within the control windrow after 8 weeks active composting ($p < 0.05$), probably largely due to a significant reduction in ammonium ions between weeks 4 and 8 (section 2.3.4).

The final EC of the treated windrow was significantly higher than the control windrow ($p < 0.05$).

Experiment 3

Table 8 shows a 3.4-fold higher level of EC in the control windrow ($p < 0.05$), due to the addition of fertiliser solution.

During 8 weeks active composting the EC level decreased $1990 \mu\text{S cm}^{-1}$ within the treated windrow ($p < 0.05$). This reflects reductions in water soluble nutrients occurred during composting (*q.v.* sections 2.3.4.c & 2.3.5).

A decrease of $181 \mu\text{S cm}^{-1}$ from initial EC was recorded for the control windrow after 8 weeks active composting ($p < 0.05$), again reflecting reductions in water soluble nutrients.

The final EC of the treated windrow was significantly lower than the control windrow ($p < 0.05$).

2.4 Discussion

Experiment 1

Moisture content was maintained in both treated and control windrows at a level generally considered not to be inhibitory to composting processes: 50-75% (Miller 1991) and >35% (Stentiford 1996).

Temperatures between 45°C and 55°C recorded within treated and control windrow over the first 4 weeks of active composting suggest that exothermic decomposition, associated with aerobic conditions, was underway. Similar temperatures have been associated with optimal respiration rates in thermophilic fungi (Gray *et al.* 1971); and fall within the optimum range of 45-60°C suggested by Bardos & Lopez-Real (1991), <60°C (Poincelot 1975).

A steep decline in temperature within the treated windrow after week 4 suggests a cessation in decomposition processes (Golueke & Diaz 1990; Haug 1993); a more gradual decline in the control windrow temperature suggests a more gradual cessation of decomposition processes.

The more rapid decline in temperature in the treated windrow concurs with the 2.3 fold greater rate of organic matter loss suggested by non-linear regression analysis of TOM content against composting duration. This indicates that the addition of water soluble macronutrients (fertiliser) significantly accelerated the decomposition of 1°WPS.

An increase in decomposition rate through the addition of supplementary N has also been observed for natural materials (Allison & Cover 1960 in Swift *et al.* 1979); and Park (1976) observed relationships between N availability and the decomposition rates of cellulose (a major component of 1°WPS organic matter). Yang *et al.* 1980 witnessed an increase in the fungal degradation of lignin [another component of WPS] from 5.2% to 29.8% on addition of 0.12%DM nitrogen to a thermomechanical pulp of alder. Where N is not a limiting factor in microbial growth, other nutrients or environmental factors increase in importance in the decomposition process (Swift *et al.* 1979).

Phosphorus is the macronutrient considered to be most important after nitrogen, in terms of the quantity required during composting. N/P ratios of between 5:1 and 20:1 for microbial cells (Alexander 1977) suggest a composting feedstock should contain phosphorus at levels between 5 and 20% of the concentration of nitrogen. Gray *et al.* (1971) suggest a C/P ratio of 75:1 to 150:1. WPS within the treated windrow possessed a starting N/P ratio of 1.94:1 and a C/P ratio of 71.2:1, suggesting P content was not a limiting factor.

Taiganides (1977) suggests the addition of 2 to 5 kg potassium chloride per tonne of composting feedstock for certain industrial wastes; equivalent to a 0.1 to 0.25%DM increase in K. A similar increase in K content (0.32%DM) was achieved here. Potassium, required at

lower concentrations than N and P, was present at a higher concentration than P within the treated windrow. It was assumed that K was present in excess; which was supported by a net mineralisation of K, *i.e.* immobilisation did not exceed mineralisation (Swift *et al.* 1979).

The water extractability (availability) of added nutrients varied considerably. Of the difference in N content between the treated and control windrows, 99% was accounted for as water soluble ammonium; 75% of added K was available in water soluble form; whereas only 17% of added P was available in water soluble form. Low levels of water soluble P in the treated windrow may be due to 'phosphate fixation', a phenomenon common in the addition of fertiliser to soil when >85% of P may be irrecoverable (Comber & Townsend 1964). Under high pH levels calcium may be the principle fixing agent (*op. cit.* 1964); and high pH and levels of calcium (calcium carbonate filler) present in WPS may have encouraged such inorganic P fixation.

During composting it appears losses of water soluble P and K were matched by losses in total P and K (when corrected for organic matter losses), suggesting nutrient leaching.

After 12 weeks composting, final TOM levels of 33.9%DM and 35.0%DM (treated and control windrows respectively) were achieved. These are broadly comparable to figures obtained for other composted materials: 36.9%DM - cattle slurry co-composted with rice hulls for 36 weeks, 8 weeks active composting (cylindrical adiabatic reactor) plus 28 weeks curing (Genevini *et al.* 1996); and 39.5%DM - a mixture of waste paper sludge, chicken litter and yard waste (8:2:1 by volume) after 27 weeks composting, 7 weeks active composting (aerated static pile) plus 20 weeks maturation (Sesay *et al.* 1997). Although TOM levels were not significantly different after 8 weeks composting plus 4 weeks maturation, there was significantly less TOM in the amended sludge after the 8 weeks composting suggesting that the nutrient addition did accelerate decomposition during the composting phase.

Total losses of TOM 30% ash and 25%ash, obtained for the treated and control windrow respectively, are lower than losses observed for other materials (composted using varying composting methods and duration): 60% - food waste and leaves (Michel *et al.* 1996); 52% cattle manure (Tarre *et al.* 1987); 75% - green waste (Frederickson *et al.* 1997); and 55% - a mixture of waste paper sludge, chicken litter and yard waste (Sesay *et al.* 1997).

Low overall TOM losses may be due to the relatively low initial TOM content of the 1°WPS (approximately 58% ash), affording less organic matter available for decomposition.

Light-microscopy revealed that paper fibres were embedded in large quantities of the inorganic matter (paper fillers and coatings), possibly forming a physical barrier to microbial attack. Gijzen *et al.* (1990) identified a similar phenomenon as a possible cause of hydrolysis

inhibition during the anaerobic digestion of a paper-mill sludge with an ash content of approximately 57%.

Although it was not quantified, the presence of lignin in paper fibres may have further restricted the biodegradability of carbon present in 1°WPS. Lignin is also thought to create a physical barrier to microbial attack (Haug 1993). Research into the biodegradability of lignin under aerobic conditions presents variable results: Lynch & Wood (1985) suggest little if any lignin degradation occurs during composting; Hammouda & Adams (1989) observed a 17% to 53% lignin degradation over 100 days during hay and straw composting; Tomati *et al.* (1995) observed a 70% reduction in the lignin content of olive waste after 23 days thermophilic composting; and Horwath *et al.* (1995) recorded a 25% and 39% lignin degradation during 45 days of grass straw composting, under mesophilic and thermophilic conditions respectively.

Ratios of cellulose and lignin vary between different WPS, *e.g.*, 23% cellulose and 16% lignin (Kannan *et al.*, 1990); 49.1% cellulose and 12% lignin for a mixture of sludges obtained from recycling and non-recycling paper mills (Butt, 1993); and 28.2-35% cellulose and 4.2-7.4% for a recycling paper-mill sludge (Gijzen *et al.*, 1990). Lignin is less generally abundant in WPS than in other lignocellulosic materials, *e.g.*, 27% lignin in green waste (Frederickson & Wood 1997).

The small (1.1%DM) difference in final TOM between treated and control windrows suggest that low macronutrient contents within the control windrow were only slightly limiting to total decomposition over 12 weeks.

The control windrow initial C/N ratio (67:1) was much higher than recommended ranges: 30:1-35:1 (Gray *et al.* 1971); 26:1-35:1 (Poincelot 1975); 30:1-50:1 (Taiganides 1977); 20:1-30:1 (Jansson & Persson 1982); 20:1-35:1 (Haug 1993). The control windrow also contained very low level of P, with a C/P ratio of 466:1. However, in terms of microbial availability, C/nutrient ratios may have been lower than those calculated from total nutrient levels; partly due to restrictions in carbon availability, discussed above.

It is also important to note that while optimal nutrient levels are often suggested, there is a considerable degree of adaptability within microbial tissues to the nutrient content in which they are growing (Swift *et al.* 1979).

A 43% loss in total nitrogen through ammonium volatilisation within the treated windrow, despite a starting C/N ratio of 37:1, further indicates an excess in available nitrogen compared with carbon. Nitrogen losses were augmented by high temperatures, ammonium concentrations and high pH (Bernal *et al.* 1993). Equilibrium kinetics suggest large increases in the dissociation of ammonium to ammonia between pH levels of 8 and 10 at 25°C (Sawyer &

McCarty 1978); if the pH of soils rise much above 8, substantial amounts of ammonia may be evolved (Richards 1974).

1°WPS was shown to be inherently basic (pH 7.9), this is postulated to be due to the presence of $\text{Ca}(\text{CO}_3)_2$, often used as a filler in the paper making process (Webb 1994). Although this was not quantified, the addition of 6M hydrochloric acid to the sludge (a test for carbonate in soils) produced the rapid and prolonged evolution of carbon dioxide gas.

The addition of nitrogen in the form of urea further increased pH to 8.9 within the treated windrow; likely to be due to the hydrolysis of urea to ammonium carbonate (Richards 1974). Jackson & Line (1997a, b) experienced similarly high pH levels and high nitrogen losses (41-62%) when composting paper mill sludge using urea as a supplementary nitrogen source.

Despite large losses in nitrogen through volatilisation during the early stages of composting, a small quantity of nitrogen (0.06%DM) was retained within the treated windrow through microbial immobilisation and via conversion into nitrate. A final level of NH_4^+ (0.003%DM) within the treated windrow after 8 weeks active composting indicates a stabilised material, falling below suggested limits of 0.04%DM (Forster *et al.* 1993, and Bernal *et al.* 1998) and 0.005%DM (Avnimelech *et al.* 1996). The production of nitrate between weeks 4 and 8 suggests favourable composting conditions (Miller 1992) and the onset of maturation (Finstain & Miller 1985). A $\text{NH}_4^+/\text{NO}_3^-$ ratio of 0.11:1 also suggested the maturity and stability of the final compost, falling below a suggested limit of 0.16:1 (Bernal *et al.* 1998).

C/N ratios achieved for the treated and control windrows, 47:1 and 53:1 respectively, are much higher than suggested levels for stabilised materials: 5:1-20:1 (Hitai *et al.* 1983), $\leq 30:1$ (ORCA 1992), $< 12:1$ (Bernal *et al.* 1998). However, a reduction in C/N ratio over 1°WPS feedstock was achieved. Grebus *et al.* (1994) suggest C/N ratio is not a good indicator of stability.

Stability indicators (NH_4^+ , NO_3^- , TOM) suggest that WPS within the treated windrow was stabilised by week 8. From TOM content it appears that stabilisation was achieved after 12 weeks within the control windrow, although low initial nitrogen levels resulted in no nitrification.

Low water extractable nutrient levels and high pH levels suggest composted WPS would provide little value as organic fertiliser; and would require amendment before it could be used as plant growth media (Rainbow & Wilson 1997). A high level of available potassium in the composted WPS obtained from the treated windrow suggest less K could be added to the initial feedstock. Conversely a greater quantity of P could be added. To reduce nitrogen losses a less alkaline nitrogen source such as ammonium sulphate or ammonium nitrate could be

investigated (Jackson & Line 1997a, b). However, pH level is affected by the composting process and is difficult to manipulate (Taiganides 1977). High pH levels of WPS have led to their suggested use as soil liming agents (Thacker 1986; and Webb 1994).

Experiment 2

Moisture content of unamended 2°WPS was very high, combined with zero porosity. The addition of straw in the treated windrow lowered initial moistures and greatly increased windrow porosity. Anaerobic conditions within the control windrow were indicated by very low windrow temperatures ($<30^{\circ}\text{C}$). Temperatures between 49°C and 57°C within the treated windrows suggested exothermic decomposition associated with aerobic conditions. High temperatures and porosity within the treated windrow resulted in rapid moisture loss, whereas the moisture content of the control windrow remained consistently above 70% during the 12 week composting period.

A steep decline in temperature within the treated windrow after week 3 suggested a rapid cessation in decomposition processes; whereas the control windrow temperature varied with ambient temperatures.

The temperature drop in the treated windrow concurred with an 85% completion of total TOM loss occurring within the first 4 weeks of composting. Proportionally, rates of TOM loss were similar for both treated and control windrow. However, total TOM loss within the treated windrow (48.8%ash) was more than double of that achieved for the control (22.9%ash).

Rapid losses of TOM in both windrows was probably due to the high availability of carbon within 2°WPS (Bellamy *et al.* 1995). Microscopic analysis revealed the 2°WPS contained very little fibre; and a C/N ratio of 9:1 suggests that organic matter in 2° WPS was likely to be in the form of microbial biomass.

Despite a higher initial TOM within the treated windrow due to the addition of straw, after 12 weeks composting the treated windrow achieved a TOM content (32.0%DM) significantly below that of the control (36.1%DM).

Nitrogen loss within the treated windrow was 43%ash, characteristic of a low initial C/N ratio (15:1). Low C/N ratios below 20:1 (Richards 1974) and 30:1 (Taiganides 1977) are known to result in nitrogen loss. Nitrogen loss was through ammonium volatilisation (encouraged by high temperatures and pH) and leaching (encouraged by high moisture content).

Losses of total nitrogen, exceeding loss of N through ammonium, suggested a net mineralisation of nitrogen and therefore a level of N in excess of microbial requirements.

Despite low temperatures, poor aeration, and lower pH levels, 25%ash nitrogen loss occurred in the control due to its very low C/N ratio (9:1).

Macronutrients P and K appear to be in adequate supply within the control and treated windrows. Although, again (see experiment 1), water soluble form of P only represents a small proportion of total P; 1.7% in the control and 4.1% in the treated windrow. Water soluble K are at greater levels than water soluble P, despite lower total concentrations. A 2-fold greater level of initial total K in the treated windrow was due to high levels of K associated with the straw amendment.

Net reductions in total P and K (corrected for DM loss) provide evidence of leachate formation in the treated windrow. No net reduction in P and K in extractable and non-extractable forms occurred within the control.

After 8 weeks composting, low NH_4^+ concentration (0.02%DM) and an increase in NO_3^- (0.03%DM) within the treated windrow suggests stabilisation has occurred. The control windrow with a 30-fold higher concentration of NH_4^+ and no NO_3^- production, suggests stabilisation did not occur, and is a further indication of anaerobic conditions. Low initial C/N ratios remained broadly unchanged for both windrows.

The final composted product derived from the treated windrow, low in water extractable P and N, would provide little value as organic fertiliser. The addition of water during the latter stages of composting (weeks 4 and 6) may have encouraged the leaching of nutrients; and as most TOM losses had already occurred this may have been an unnecessary procedure.

High levels of water extractable K suggest the compost would require the balancing of macronutrients before use as plant growth media (Rainbow & Wilson 1997). The utilisation of a bulking agent with a lower K content may alleviate this.

A low final C/N ratio suggests that the land application of this compost would not result in nitrogen immobilisation, and complies with C/N ranges suggested for stabilised material: 5:1-20:1 (Hitai *et al.* 1983), $\leq 30:1$ (ORCA 1992).

Composting 2°WPS with straw greatly reduced odour and moisture levels, improving handling properties and reducing the potential for environmental impact, such as odour and leachate production.

Experiment 3

A 2:1 (by volume) mixture of 1°WPS and 2°WPS created a substrate with physicochemical characteristics broadly comparable to that achieved by the addition of fertiliser to 1°WPS in experiment 1.

Moisture contents were maintained in both treated and control windrows at a level of between 45% and 55%. Temperatures between 45°C and 52°C recorded within treated and control windrows over the first 3 weeks of active composting suggest exothermic decomposition associated with aerobic conditions.

A steep decline in temperature within the control windrow after week 3 suggested a cessation in decomposition processes; whereas temperatures within the treated windrow did not decrease until after week 7.

A more rapid decline in temperature concurs with the 3.6-fold greater rate of organic matter loss in the control windrow suggested by non-linear regression analysis of TOM contents. The addition of nutrients in the form of fertiliser to the treated windrow mixture appears to have inhibited TOM degradation rate.

Non-linear regression also showed that the rate of TOM loss and the total TOM loss in the control were similar to those observed for 1°WPS amended with fertiliser (experiment 1). Final TOM contents of 33.5%DM achieved in both treated and control windrows were similar to those achieved in treated windrows of experiments 1 and 2.

Total nitrogen losses of 30%ash and 26%ash for treated and control windrows, were lower than observed for 1°WPS plus fertiliser, despite possessing higher initial total N contents.

C/N ratios showed little change during composting, and final levels were broadly similar between control and treated levels. Final C/N ratios of 33:1 (control) and 31:1 (treated) are above levels recommended for stabilised materials.

Of the nitrogen retained, little remained in the form of NH_4^+ , 0.008%DM (control) and 0.058%DM (treated). The treated windrow contained NH_4^+ above limits suggested for stabilised materials of 0.04%DM (Forster *et al.* 1993, and Bernal *et al.* 1998) and 0.005%DM (Avnimelech *et al.* 1996).

The formation of NO_3^- between weeks 4 and 8 indicated favourable composting conditions (Miller 1992) and the onset of maturation (Finstein & Miller 1985). $\text{NH}_4^+/\text{NO}_3^-$ ratios of 0.58:1 (control) and 3.78:1 (treated) suggest a greater maturity and stability of the final compost derived from the control windrow, although both are above a suggested limit of 0.16:1 (Bernal *et al.* 1998).

Reductions in water soluble P and K during composting appeared to be due to microbial immobilisation. No overall loss in total P and K suggest leaching of those nutrients did not occur. Again (see experiments 1 and 2) water extractable P formed only a small fraction of the total P content. The addition of fertiliser greatly increased the final value of water extractable K in the treated windrow, a phenomenon observed in experiment 1.

The combination of 1° and 2°WPS appears to have produced a stabilised material after 8 weeks composting, with a similar progression in TOM loss observed for 1°WPS after the addition of fertilizer. Lower nitrogen loss and P and K leaching suggest 2°WPS may provide macronutrients in a form more easily retained than inorganic fertilizers (experiment 1). The effects on the final composted product appear to be a lower pH, lower C/N ratio and better balance between K and other macronutrients, than observed for WPS amended with fertilizer (experiment 1).

The addition of fertilizer to the mixture of 1° and 2°WPS proved detrimental to the composting process, in terms of a slower TOM degradation rate and increased nitrogen loss. Inhibition of TOM degradation may have been caused by excessive production of ammonia gas from a combination of initially high ammonium levels (0.48%DM) combined with a high pH (8.8).

The addition of fertiliser did not appear to greatly affect the final composted product, other than provide an excess of water extractable K (*q.v.* experiment 1).

Combining 1° and 2°WPS provided a means of stabilising both sludges simultaneously, again reducing odours and improving the handling properties of 2°WPS. As both these products are produced by papermills, composting could be carried out without the cost of acquiring other amendments.

2.5 Conclusions

The main aim of this Chapter was to investigate the stabilisation of two types of waste paper sludge using readily available windrow composting technology. In order to minimise intervention and to facilitate ease of adoption by the paper making industry, both of the selected sludges (primary and secondary) were only minimally amended to allow effective composting to take place.

In experiment 1, physicochemical analysis clearly showed that the selected primary sludge had a very high C/N ratio (67:1) and amendment with major nutrients would be required. This was achieved through the addition of soluble fertiliser. As a result of composting and maturation for 12 weeks, the nutrient-amended sludge, lost 30% of its organic matter while the unamended sludge lost 25% organic matter, based on ash content, which was not significantly different. The nutrient-amended sludge also lost much of its total nitrogen through volatilisation (43%) during the first four weeks of composting and this resulted in an increase in C/N ratio from the initial 37:1 to 47:1 after maturation. Despite this increase, the C/N ratio of the amended sludge was still significantly less than the non-amended sludge after composting, suggesting that amendment was useful in terms of producing a stabilised compost with a lower C/N ratio.

Although TOM levels were not significantly different after 8 weeks composting plus 4 weeks maturation, there was significantly less TOM in the amended sludge after the 8 weeks composting, suggesting that the nutrient addition did accelerate decomposition during the composting phase. In addition, other stability indicators (NH_4^+ , NO_3^-) suggest that WPS within the treated windrow was stabilised by week 8. It would appear that there may be scope for accelerating the decomposition of primary sludge by amending with nutrients but on balance the benefits may not be cost-effective in practice.

In experiment 2 a bulking agent was added to the wet, low C/N ratio secondary sludge in order to increase porosity and act as a carbon source. The untreated windrow (control) was the unamended sludge. During composting, a steep decline in temperature within the treated windrow after week 3 suggested a rapid cessation in decomposition processes; whereas the control windrow temperature varied with ambient temperatures throughout, confirming that composting was not taking place. The temperature drop in the treated windrow concurred with an 85% completion of total TOM loss occurring within the first 4 weeks of composting. However, total TOM loss within the treated windrow (48.8%ash) was more than double of that achieved for the control (22.9%ash). Despite a higher initial TOM within the treated windrow due to the addition of straw, after 12 weeks composting the treated windrow achieved a TOM content (32.0%DM) significantly below that of the control (36.1%DM).

Nitrogen loss within the treated windrow was 43%ash, characteristic of a low initial C/N ratio (15:1). Nitrogen loss was through ammonium volatilisation (encouraged by high temperatures and pH) and leaching (encouraged by high moisture content). Despite low temperatures, poor aeration, and lower pH levels, 25%ash nitrogen loss occurred in the control due to its very low C/N ratio (9:1). After 8 weeks composting, a low NH_4^+ concentration (0.02%DM) and an increase in NO_3^- (0.03%DM) within the treated windrow were noted, suggesting that stabilisation has occurred. The control windrow with a 30-fold higher concentration of NH_4^+ and no NO_3^- production, suggests stabilisation did not occur.

Without the addition of the straw bulking agent, effective composting did not take place whereas amending the secondary sludge with straw promoted vigorous composting and rapid stabilisation.

In experiment 3 a mixture of 1°WPS and 2°WPS (2:1 by volume) created a substrate with physicochemical characteristics broadly comparable to that achieved by the addition of fertiliser to 1°WPS in experiment 1. The treated windrow contained additional fertiliser compared with the control windrow.

Temperatures between 45°C and 52°C were recorded within treated and control windrows over the first 3 weeks of active composting but a steep decline in temperature within the control windrow thereafter suggested a cessation in the decomposition processes. Temperatures within the treated windrow did not decrease until after week 7. A more rapid decline in temperature concurs with the 3.6-fold greater rate of organic matter loss in the control windrow. The addition of nutrients in the form of fertiliser to the treated windrow mixture appears to have inhibited TOM degradation rate.

The rate of TOM loss and the total TOM loss in the control windrow were similar to those observed for 1°WPS amended with fertiliser (experiment 1). Final TOM contents of 33.5%DM achieved in both treated and control windrows were similar to those achieved in treated windrows of experiments 1 and 2.

C/N ratios showed little change during composting, and final levels were broadly similar between control and treated levels. Final C/N ratios of 33:1 (control) and 31:1 (treated) are above levels recommended for stabilised materials. The formation of NO_3^- between weeks 4 and 8 indicated the onset of maturation suggest a greater maturity and stability of the final compost derived from the control windrow.

The combination of 1° and 2°WPS appears to have produced a stabilised material after 8 weeks composting but the addition of fertilizer to the mixture of 1° and 2°WPS proved detrimental to the composting process, in terms of a slower TOM degradation rate and

increased nitrogen loss. Combining 1° and 2°WPS provide a means of stabilising both sludges simultaneously and since both wastes are produced by papermills, composting could be carried out without the cost of acquiring other amendments.

3. Earthworm population sustainability and growth

3.1 Introduction

Experiences with primary and secondary WPS derived from paper recycling mills, though amenable to windrow composting (Chapter 2), revealed they were unsuitable for vermicomposting (Appendix 2). A pulp derived from a mixture of sludges produced as by-products of primary and secondary waste water treatment (Appendix 1) provided the desired nutritional characteristics, and was therefore investigated. The sludge was obtained from a board recycling mill (SCA Packaging Ltd.). The vermicomposting process results in two products of potential economic value: worm mass and vermicompost, as well as reducing waste treatment costs (Foote & Fieldson 1982 and Fieldson 1988). However, during these analyses the costs of waste disposal, animal manure in this case, were lower and less restricted by legislation than paper sludge is today. These factors make the economics of vermicomposting more viable than earlier economic analysis may have revealed (Chapter 1).

The maintenance of an adequate population of earthworms is essential to the vermicomposting process, both in terms of worm production and waste processing (vermicompost production). However, the culturing conditions required to optimise these two aspects are antagonistic. Edwards (1988) suggested that for mass worm production, cocoons or young worms should be used, whereas for vermicompost production almost fully-grown worms should be used.

Earthworm density during vermicomposting is a vitally important parameter and studies into the effects of stocking density upon worm reproduction and growth have been conducted on various species (Neuhauser *et al.* 1980a,b; Reinecke & Viljoen 1990; Butt *et al.* 1994; Dominguez & Edwards 1997). Few have been conducted with *Dendrobaena veneta* [the species employed in this study] (*e.g.* Reeh 1992). This is not surprising considering its lower reproductive potential compared to species such as *Eisenia fetida/andrei* (*e.g.* Edwards 1988). Fish-bait is one of the largest international markets for worms (Tomlin 1983). However, it has been observed that *E. fetida* does not make good fishing bait (Knight 1989) and feeding trials have shown *D. veneta* to be much more acceptable to rainbow trout than *E. fetida* (Stafford & Tacon 1988). *D. veneta* has also been shown to be one of the most efficient epigeic (soil surface dwelling) species for converting waste into worm biomass, as well as producing the largest individual worm mass [another characteristic desired by anglers, Tomlin (1983)] (Neuhauser *et al.* 1988).

The effect of worm density on waste processing rates has been little documented, despite the higher potential economic value of vermicompost than worms (Foote & Fieldson 1982). In recent years there has been a trend towards vermicomposting sewage and industrial sludges and the effect of various metals on worm growth. Reproduction has also been investigated to a limited extent (Edwards & Bohlen 1992). However, despite its toxicity to worms, few studies have investigated aluminium, which is the dominant metal found in many sludges from the paper making industry (Philips & Bolger 1998).

The overall objective of this Chapter is to evaluate the suitability of WPS for vermicomposting with particular reference to the effect of WPS on worm survival, growth and reproduction.

The aims of this Chapter are:

- 1) To investigate the effect of WPS on worm survival,
- 2) To investigate the nutritive value of WPS as a feed for the earthworm *D. veneta*,
- 3) To investigate the effect of stocking density on the growth (conversion of WPS to worm mass) and reproduction of *D. veneta*,
- 4) To investigate the effect of stocking density on the rate and degree of WPS processing by *D. veneta*.

3.2 *Materials and methods*

3.2.1 Experimental earthworm populations and WPS feedstock

Earthworms

Juvenile (pre-clitellate) worms were obtained from a stock culture of the epigeic earthworm species *Dendrobaena veneta*. Worms were hand-sorted into two classes, small juveniles (140 ± 11 mg WM; mean \pm se) and large juveniles (825 ± 48 mg wet mass; mean \pm se). Each experimental stock population was normally distributed in terms of wet-mass ($p > 0.1$; Kolmogorov and Smirnov test).

WPS feedstock

Two batches of fresh WPS feedstock were used (A and B) obtained from SCA Packaging Ltd. Batch A was used over the first half of the experimental period (weeks 0 to 4), and batch B over the second half (weeks 4 to 8). Each batch was stored at between 0 and 4°C for a maximum of 2 weeks to prevent excessive degradation.

WPS feed moisture contents of 79.6% (batch A) and 74.0% (batch B) were within acceptable levels (*q.v.* Muiyima *et al.* 1994) and were therefore not adjusted.

3.2.2 Effect of worm density upon worm mass production, survival, reproduction, and WPS processing rate

Experimental design

For each experimental replicate, either small or large juvenile worms were assigned randomly to 0.5 litre (120-mm diameter) cylindrical, clear, plastic pots with ventilated lids. Stocking cultures were set at densities of: 1 (N=10), 2 (N=10), 4 (N=10), 8 (N=5) and 16 (N=5) worms pot⁻¹. Each replicate was supplied with 100g of fresh waste-paper-pulp (WPS), which was replenished every 2 weeks. All 40 experimental pots were incubated in an air-conditioned incubator (SANYO Gallenkamp, model ICX180.XX2.C) at 20°C (*q.v.* Viljoen *et al.* 1992 and Fayolle *et al.* 1997).

Data collection and analysis

Nutritional composition of fresh WPS

To assess the nutritional content of the fresh WPS-feed, total-organic-carbon (TOC), total-nitrogen (N), total-phosphorous (P), and total-potassium (K), total-calcium (Ca) as well as water extractable forms of N, P and K, Mg and Ca were determined (see Physicochemical analyses, below).

Worm mass production

Earthworms were weighed (mg wet-mass) every week over an eight-week period. High levels of earthworm mortality were experienced at certain densities making statistical comparisons difficult. To overcome this, non-linear regression analysis was employed. Curves were fitted using each replicate as an individual data point, and the mean (\bar{x}), standard error (se) and degrees of freedom (df) were calculated for each variable of the curve-of-best-fit (equation 1).

$$Y \text{ (increase in worm mass)} = a \times (1 - e^{(-bX)}) + c \quad (1)$$

a = total increase in worm mass; b = constant defining the rate at which maximum worm mass was reached; c = initial worm mass (fixed as zero).

Curves were fitted using GraphPad Prism[®] statistical analysis software package (© GraphPad Software Inc.), using the Levenberg-Marquardt method of estimation (sum-of-squares minimisation).

The model provided scientific assumptions felt appropriate for modelling worm mass increase, *i.e.*, a decelerating rate of increase before reaching a constant maximum value (within the experimental period).

Variables [a and b] of curves of best fit can be compared with Student t-tests, using \bar{x} , se, and N (df+1), *q.v.* Motulsky (1996). Therefore, statistical differences between curve variables, derived from different stocking density treatments, were calculated using ANOVA and post-ANOVA, Tukey-Kramer multiple comparison tests.

Worm Survival

Total earthworms were counted every week for each density treatment. Each absent or dead worm was recorded as a 'survival' event, $y=1$ (Motulsky 1995). After 8 weeks each surviving worm was 'censored', $y=0$ (*op. cit.*). Data were plotted as survival curves, and curves were compared, statistically, using log-rank tests (*q.v. op. cit.*).

Reproduction

The time taken for clitellum development is not always proportional to the onset of cocoon production in earthworms (*e.g.*, *Eisenia foetida*, Hartenstein 1981). Therefore, the onset and rate of cocoon production were used to measure reproduction.

The cocoons produced from each stocking density treatment were collected and counted every week. To evaluate their viability and levels of polyembryony (number of hatchlings per

cocoon), cocoons collected from different weeks were amalgamated and incubated at 20°C on filter paper saturated with distilled-water inside petri dishes.

Hatched cocoons were counted every week of incubation, and each hatched cocoon recorded as a 'survival' event, $y=1$ (Motulsky 1995). Each unhatched cocoon was 'censored', $y=0$ (*op. cit.*). Data were plotted as survival curves and curves compared statistically using log-rank tests (*q.v. op. cit.*).

WPS processing (worm egesta [casts] production)

Unprocessed WPS-feed, and WPS processed into egesta (casts) by large juvenile worms, was removed every two weeks. This material was weighed, dried at 105°C for 24 hours in a fan-assisted oven (Haraeus Instruments, model UT-6200), and sieved into two fractions: >3.35 mm (unprocessed WPS) and <3.35 mm (WPS casts).

To evaluate organic matter losses affected by worm processing, the total organic matter (TOM) content of processed-WPS (casts) and unprocessed WPS were determined (see Physicochemical analyses below).

Conversion ratio

The conversion of WPS into worm mass was calculated every two weeks for juvenile worms; firstly in terms of the cumulative wet mass of WPS-feed supplied¹, and secondly in terms of the cumulative wet mass of WPS processed². Cumulative increases in worm wet-mass were calculated as the summation of the differences in worm mass between the beginning and end of each 2-week period.

Aluminium uptake by earthworms

To assess the potential uptake of aluminium by worms during WPS processing, the total aluminium contents were determined for the WPS feedstocks and for single composite samples of worms, derived from both the small and large juveniles after culture upon WPS for eight weeks.

Earthworms cultured on WPS for eight weeks were separated from the WPS substrate and allowed to gut-void for 24 hours in distilled water. They were killed by freezing at -18°C, and dried in a desiccating oven at 105°C for 24 hours. Dried, milled (<1mm) samples were sent to an external lab for analysis (see Physicochemical analyses below).

¹ Calculated for small and large juveniles; ² Calculated for large juveniles only

Total organic matter (TOM) and total organic carbon (TOC)

Total organic matter (volatile solids) content was determined as the loss in dry-mass upon ignition at 550°C in a 2.5-litre muffle-furnace (SANYO Gallenkamp PLC, model FRM002.XX2.5) for 2 hours (FCQAO 1994). Samples were previously dried at 105°C for 24 hours in a fan-assisted desiccating oven (Haraeus Instruments, model UT-6200).

The total organic carbon content (TOC) was determined as total organic matter (volatile solids) / 1.8 (Adams *et al.* 1951).

Mineral nutrients: nitrogen (N), phosphorus (P) and potassium (K)

Total N, P, K, Ca contents, the content of water-extractable forms of N [NO₃⁻] & [NH₄⁺], P [H₂PO₄⁻], K [K₂O], Mg²⁺, Ca²⁺, pH and electrical conductivity (EC) were determined. Determinations were conducted by Levington Agriculture Ltd (Levington Park, Ipswich, Suffolk, UK); using standard procedures.

Total aluminium (Al) content

The total aluminium contents of WPS and worms were determined by ADAS[®] laboratories (Woodthorne, Wergs Road, Wolverhampton, UK) using standard procedures.

3.3 Results

3.3.1 Nutritional composition of fresh WPS

Table 9 – Physicochemical analysis of fresh WPS feedstocks

WPS feedstock	TS ¹ %	TOM ² %DM	Fibre content (%DM)			Total nutrients (% DM)					
			Cellulose	Hemi-cellulose	Lignin	C	N	C:N	P	K	Ca
Batch A	20.4	84.4	32	14	8	46.4	2.75	16.9	0.45	0.23	2.02
Batch B	26.0	85.3				46.9	2.11	22.2	0.38	0.23	1.79
Difference of A-B (% of A)	27%	1.1%	–	–	–	1.1%	-23%	31%	-16%	0%	-11%

Continued

WPS feedstock	Water-extractable nutrients (mg kg ⁻¹ WM)						Ionic factors	
	NH ₄ ⁺	NO ₃ ⁻	P	K	Mg	Ca	PH	EC ³
Batch A	274	8	71	265	83	2268	6.0	1360
Batch B	148	6	44	264	71	1864	6.0	984
Difference of A-B (% of A)	-46%	-25%	-38%	-0.4%	-15%	-18%	–	-28%

¹: Total solids – w/w; ²: Total organic matter content; ³: Electrical conductivity

TOM (carbon) contents of WPS batches was found to differ by only 1% (Table 9). TOM was present mainly in the form of fibres composed of 34% cellulose, 14% hemi-cellulose and 8% lignin.

Batch A WPS (used up to week 4) was higher in total and water extractable mineral nutrients than batch B (used up to week 8). A 23% lower total nitrogen content in batch B was partly reflected in lower water extractable ammonium (46%), nitrate (25%) contents, and a higher C:N ratio. Water extractable nutrients (P, K, Mg and Ca) were also lower within batch B WPS, with a correspondingly lower EC.

3.3.2 Worm growth

Total worm mass production

Figure 10 – Increase in the total worm mass; small juveniles reared at stocking densities of: 16: ●—, 8: *—, 4: X—, 2: +—, 1: ○----

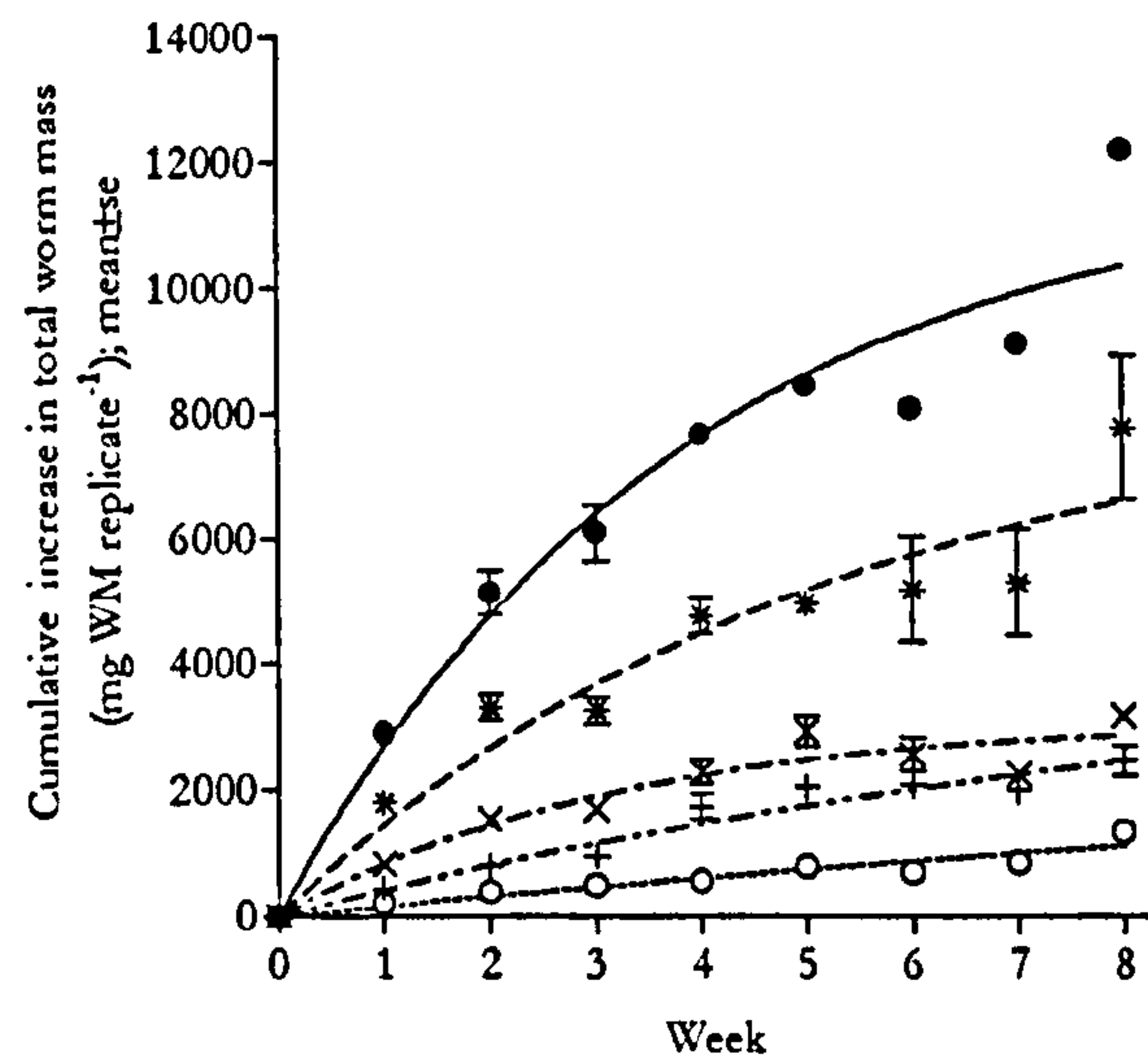


Figure 11 – Increase in total worm mass; large juveniles reared at stocking densities of: 16: ●—, 8: *—, 4: X—, 2: +—

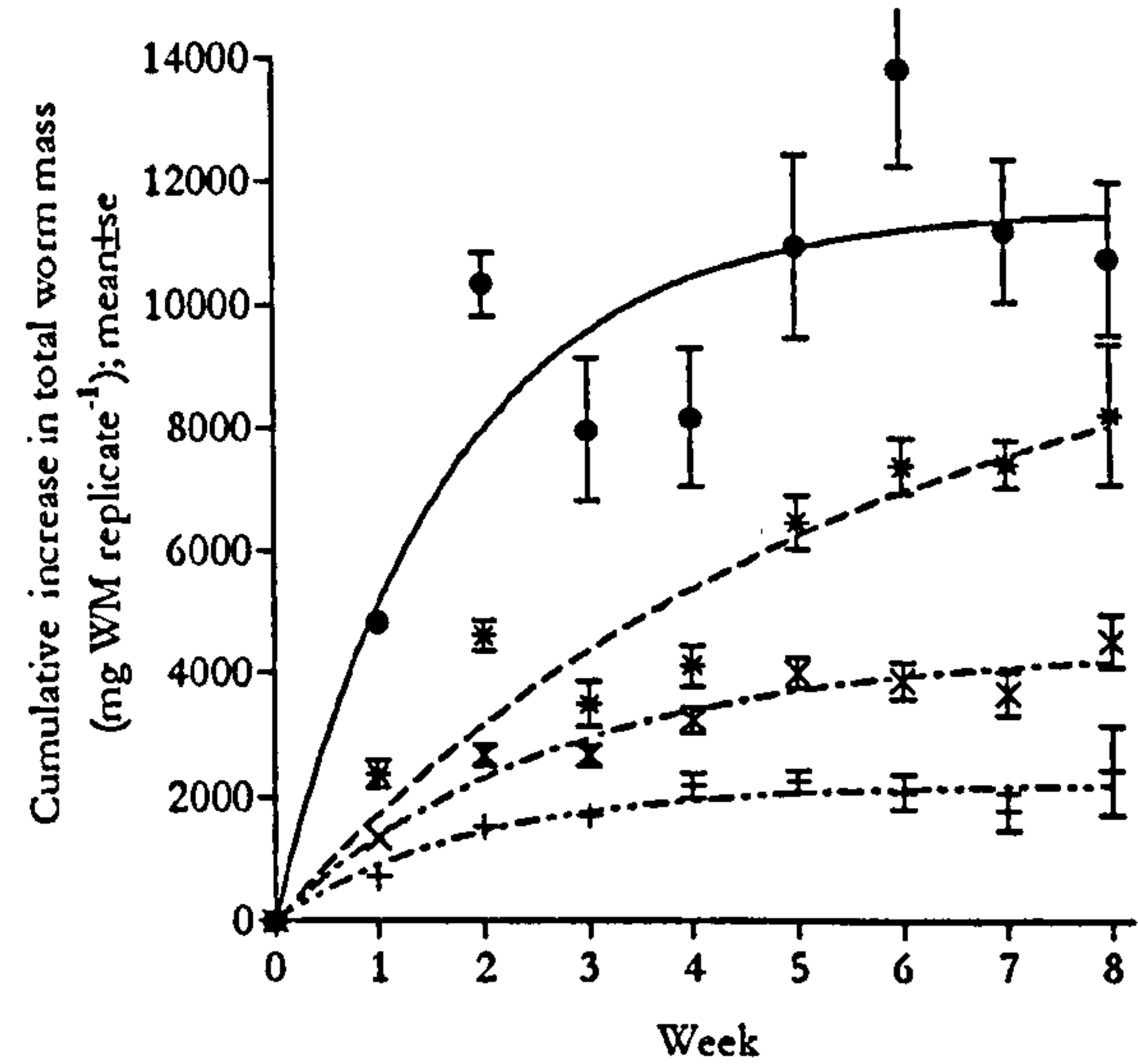


Table 10 – Non-linear regression results

Small juveniles					Large juveniles				
Curve variable	Stocking density (no. of worms)				Curve variable	Stocking density (no. of worms)			
	16	8	4	2		16	8	4	2
A	11.90±110 ^a	8.47±1.68 ^a	3.13±0.22 ^b	4.57±1.16 ^b	A	11.55±0.80 ^a	10.60±1.93 ^a	4.41±0.27 ^b	2.23±0.16 ^c
B	0.26±0.04 ^a	0.19±0.06 ^a	0.31±0.05 ^a	0.10±0.03 ^a	B	0.59±0.15 ^a	0.18±0.06 ^a	0.37±0.06 ^a	0.52±0.12 ^a
Measures of goodness of fit					Measures of goodness of fit				
R ²	94.5%	77.2%	81.3%	89.3%	R ²	66.0%	79.5%	73.7%	58.4%
S _{x,y}	0.78	1.15	0.46	0.27	S _{x,y}	2.66	1.27	0.80	0.63

abc : Different letters indicate significant differences (p<0.05, Tukey–Kramer multiple comparisons test)
R² : Fraction of total variance in experimental data explained by the model (Motulsky 1996)
S_{x,y} : Standard deviation of points from the model line-of-best-fit (*op. cit.*)

Total worm mass production (variable a, non-linear regression)

All worms showed significant increases in mass over the 8 week experimental period (p<0.05, Student t-test).

Generally, it was observed that increased stocking density had an incremental effect upon total worm mass production when rearing either small or large juveniles for 8 weeks, predicted by non-linear regression (Figure 10, Figure 11, and Table 10).

Earthworm density was found to be significantly related to total biomass production. ANOVA post test for linear trend², showed a significant relationship between decreasing

² The ANOVA post-test for linear trend determines whether the column means increase (or decrease) systematically as the columns go from left to right (Table 10 – densities of 16 to 2 worms).

stocking density and the reduction in total worm mass production for small juveniles: slope = -1.37 g per sequential decrease in stocking density; $p < 0.0001$; $R^2 = 11.5\%$. Results for large juveniles showed a greater reduction in total worm mass production with decreasing stocking density: slope = -1.71 g per sequential decrease in stocking density; $p < 0.0001$; $R^2 = 25.8\%$.

However, both trends showed significant levels of non-linear variation ($p < 0.05$ and $p < 0.01$, respectively). Therefore, total worm mass production values were pooled and plotted against the stocking density of worms in terms of number (Figure 12).

Figure 12 – Changes in total worm mass increase with increasing stocking density; line of best fit used for graphical purposes only.

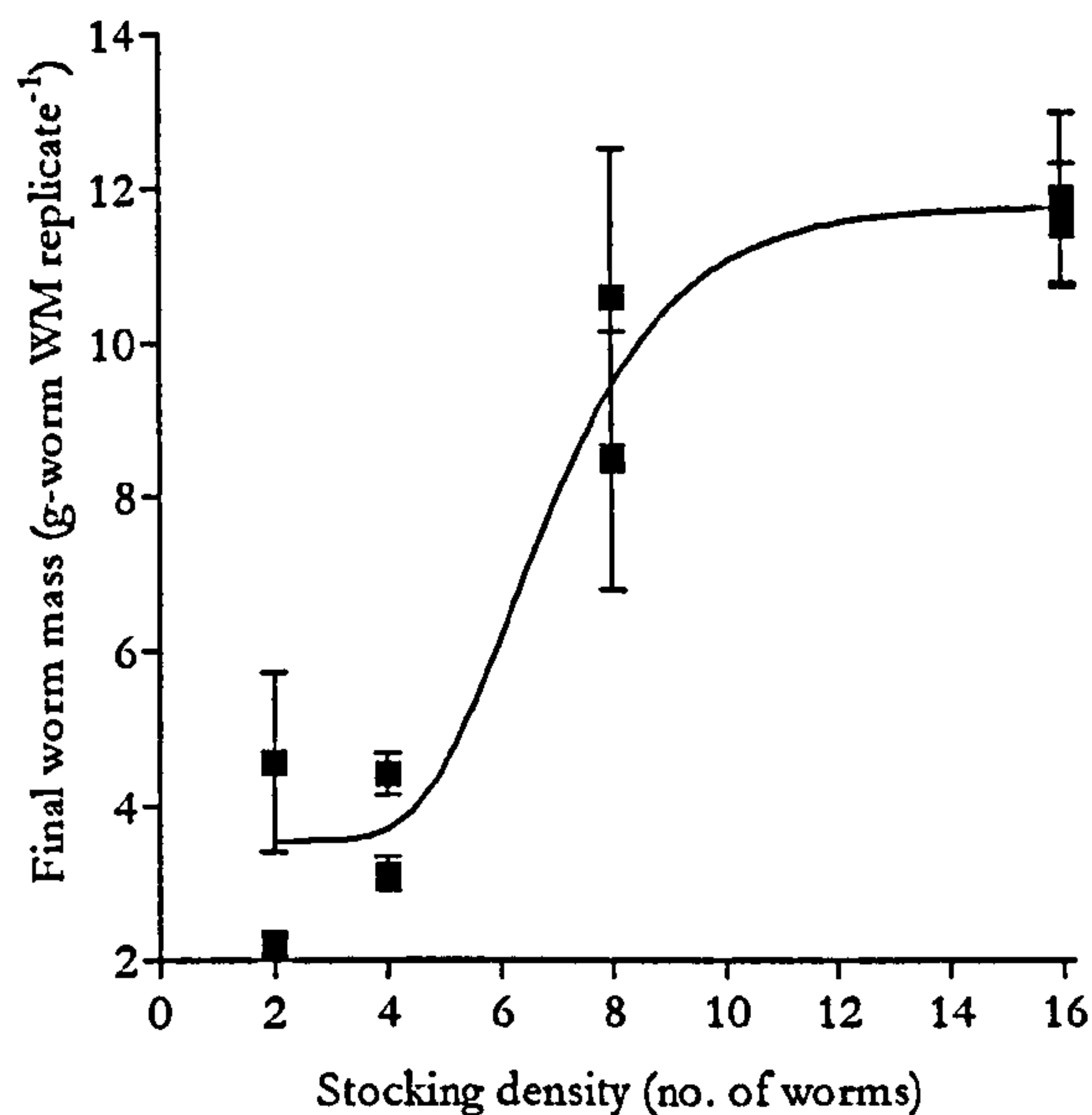


Figure 12 shows that increased initial worm mass caused a steep increase in total worm mass production between densities of 2/4 worms (2.23 to 4.57g) and 8/16 worms (8.47 to 11.90g). This was supported by statistically significant differences between all mean total worm mass production values of 2/4 and 8/16 worms ($p < 0.05$, Tukey-Kramer multiple comparisons test). Differences in total worm mass production between densities of 2 and 4 worms, and 8 and 16 worms, were not significantly different ($p > 0.05$, Tukey-Kramer multiple comparisons test).

Significant differences between the total worm mass production of small juveniles and large juveniles were only observed at lower worm densities (2 and 4 worms). At a stocking density of 2 worms, small juveniles produced significantly greater total worm mass than large juveniles over 8 weeks ($p < 0.05$, Tukey-Kramer multiple comparisons test). Conversely, at a density of 4 worms, large juveniles produced a significantly greater total worm mass than small juveniles over 8 weeks ($p < 0.05$, Tukey-Kramer multiple comparisons test).

Rate at which total worm mass production was obtained (variable b, non-linear regression)

Differences between the rate at which worm growth curves reached a plateau (maximum worm mass) were not statistically significant ($p>0.05$, one-way ANOVA), largely due to large standard errors (16 to 33% of the mean).

However, the rates at which maximum levels of total worm mass were reached were generally higher for large juvenile worms. Higher rates at which worm growth ceased at high stocking densities may also indicate smaller final worm masses, due to restricted growth. This was reflected in differences between the final mass of individual worms reared at different densities (Table 11).

Individual worm growth

Table 11 – Total Individual worm growth after 8 weeks

Stocking Density (no. of worms)	Small juveniles (after 8 weeks)		Large juveniles (after 8 weeks)	
	Final individual mass mg*	Growth rate† mg* worm ⁻¹ day ⁻¹	Final individual mass mg*	Growth rate mg* worm ⁻¹ day ⁻¹
2	1383±121 ^a	22.3±2.1 ^a	2091±320 ^b	21.4±5.7 ^a
4	938±28 ^a	14.1±0.6 ^a	1933±99 ^b	20.3±1.7 ^b
8	1123±146 ^a	17.4±2.6 ^a	1834±145 ^b	18.4±2.6 ^a
16	888 (N=1)† ^a	13.7 (N=1)† ^a	1492±70 ^b	12.1±1.3 ^a

† : Growth rates were calculated from final individual worm increases divided by 56 days (8 weeks).
* : All values presented in terms of wet-mass
abc : Different letters indicate significant differences between small and large juveniles ($p<0.05$, Student t-tests with Welch's correction)
‡ : This value was used as the hypothetical mean in a one-sample Student t-test between columns

Generally, increased stocking density resulted in a decreased individual mass of worms reared over 8 weeks (Table 11). However, no statistical differences were observed, again, possibly due to large levels of variation (se) and reduced replicates (N) due to worm high mortality.

Whereas the final mass of large juveniles decreased systematically with increased stocking density, small juveniles showed erratic levels of final worm mass. The final individual mass of worms obtained from a stocking density of 4 small juveniles (938 mg), was lower than that obtained with 8 small juvenile worms (1123 mg), and only 50 mg greater than that observed for 16 small juveniles (888 mg).

Large juvenile worms showed significantly greater final individual masses than small juveniles at all densities ($p<0.05$). This suggested that cessation in worm growth within small juveniles cultures were unlikely to be due to worms reaching maximum obtainable individual worm mass.

Individual growth rates, as expected, also decreased with increased stocking density. However, growth rates were broadly similar ($P>0.05$) between small and large juveniles cultured at similar densities (no. of worms). A density of 2 worms produced mean individual

growth rates of 21.4-22.3 mg worm⁻¹ d⁻¹; 4 worms – 14.1-20.3; 8 worms – 17.4-18.4; and 16 worms – 12.1-13.7 mg worm⁻¹ d⁻¹.

3.3.3 Conversion ratios

Figure 13 – Total worm production % total-WPS feed; small juveniles reared at stocking densities: 16: ●—; 8: *—, 4: X—

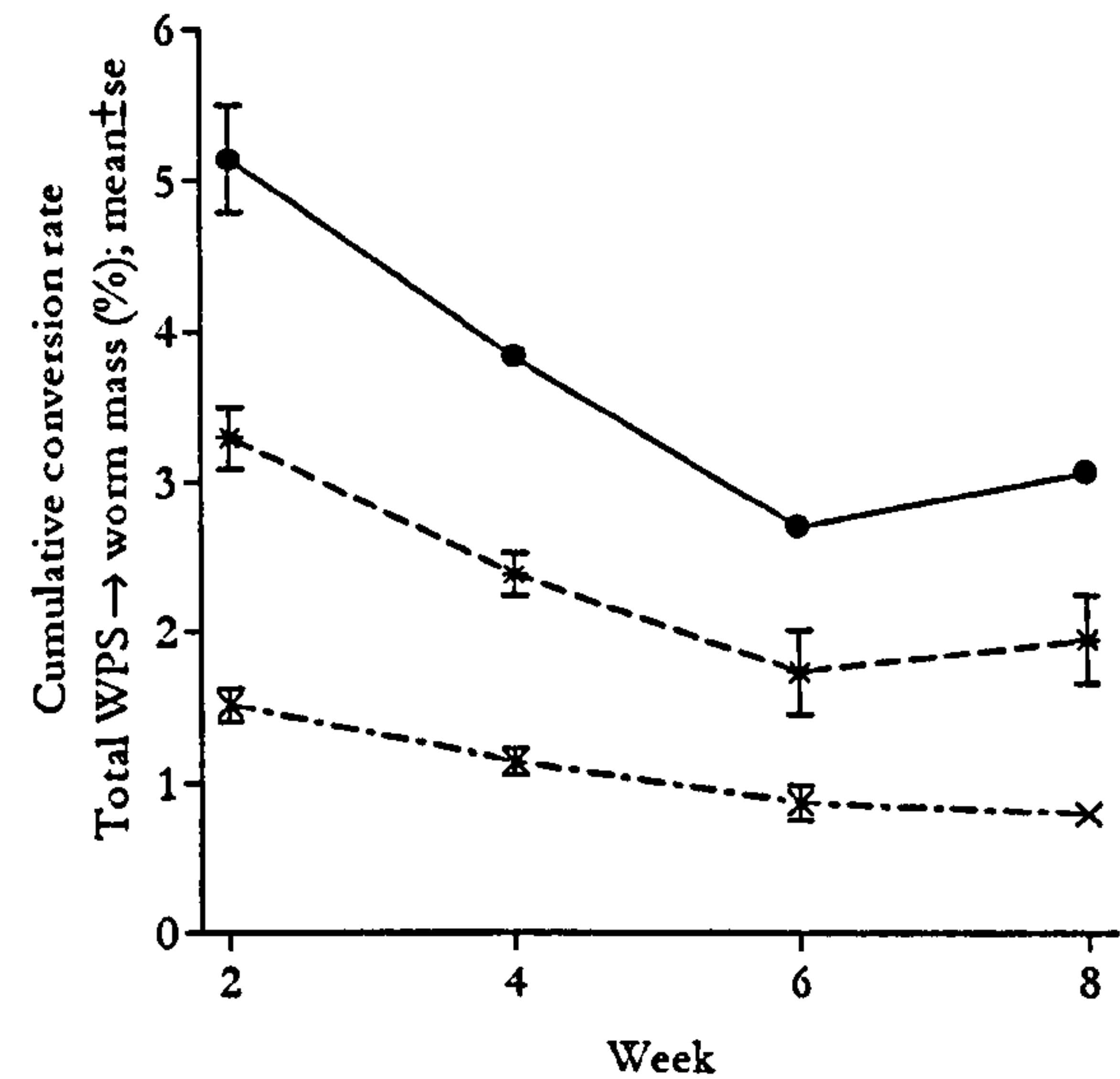


Figure 14 – Total worm production % total-WPS feed; large juveniles reared at stocking densities: 16: ●—; 8: *—, 4: X—

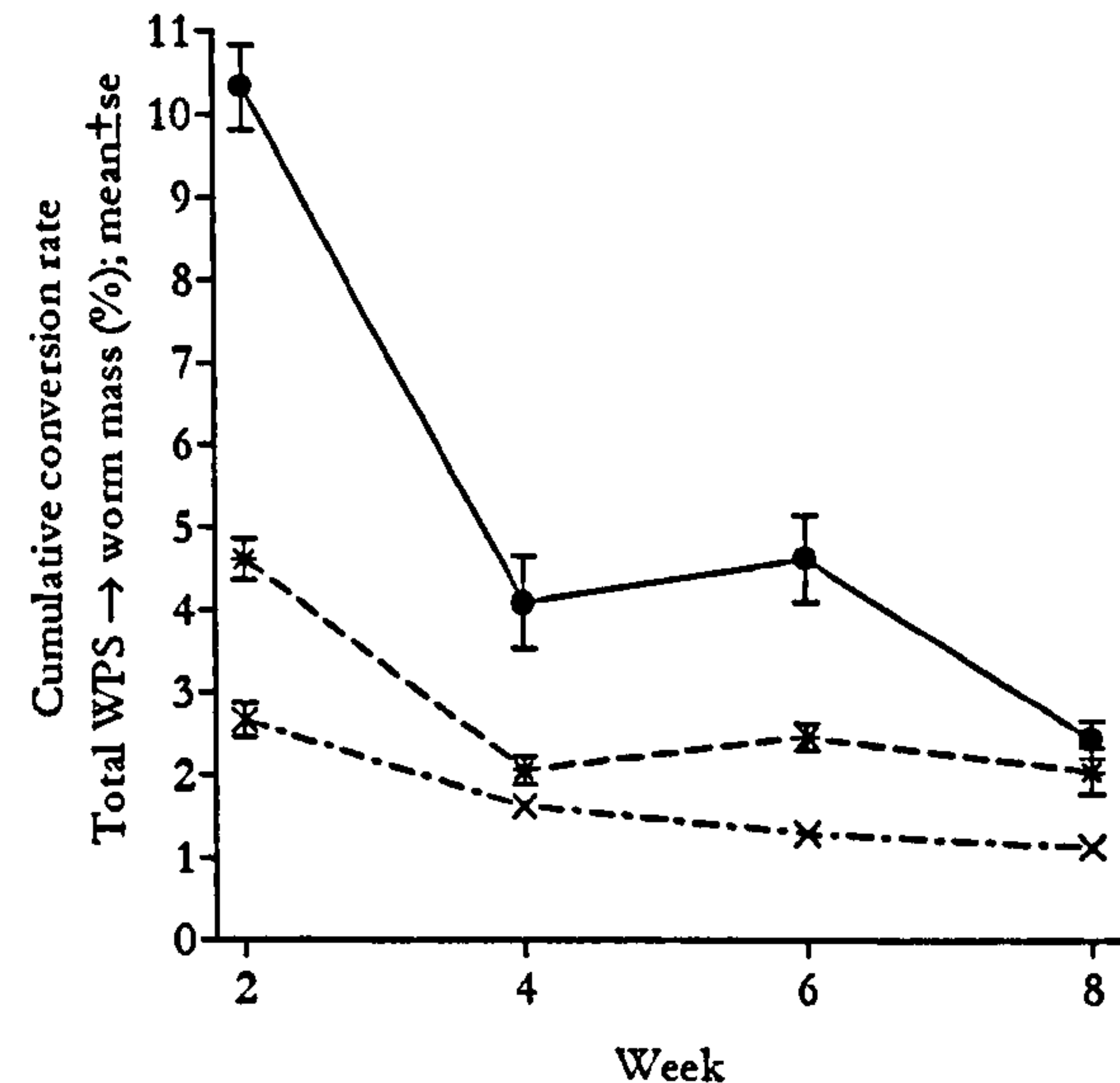


Table 12 – Cumulative conversion of WPS-feed into worm mass (%); mean±se

Small juveniles					Large juveniles				
Week	Stocking density				Week	Stocking density			
	ANOVA	16	8	4		ANOVA	16	8	4
2	***	5.14±0.36 ^a	3.29±0.20 ^b	1.52±0.11 ^c	2	***	10.33±0.51 ^a	4.61±0.24 ^b	2.66±0.21 ^c
4	t-test ***	3.83 (N=1)	2.38±0.14 ^a	1.14±0.08 ^b	4	***	4.09±0.56 ^a	2.06±0.17 ^b	1.62±0.12 ^b
6	t-test *	2.69 (N=1)	1.73±0.28 ^a	0.87±0.12 ^b	6	***	4.62±0.53 ^a	2.46±0.15 ^b	1.29±0.11 ^b
8	t-test ^w *	3.06 (N=1)	1.95±0.28 ^a	0.79±0.02 ^b	8	**	2.44±0.22 ^a	2.06±0.28 ^{ab}	1.13±0.12 ^b

One-way ANOVA and unpaired t-tests: * p<0.05; ** p<0.01; *** p<0.001;
abc : Different letters indicate significant differences (p<0.05, Tukey–Kramer multiple comparisons test)
^w : t-test with Welch’s correction

The conversion of total WPS-feed into worm mass was higher for higher stocking densities (Figure 13, Figure 14 and Table 12), coinciding with higher total worm mass production at higher stocking densities (section 3.3.2).

Greater cumulative conversion ratios over the first 2 weeks of the experimental period were observed, generally decreasing with each subsequent 2-week period. This was consistent with decreasing worm growth as maximal worm masses were reached (section 3.3.2).

Density had a significant effect on conversion during each 2-week feeding period (p<0.05, one-way ANOVA, Tukey-Kramer multiple comparisons tests and t-tests). During the first 2-week period, for either small or large juveniles, higher mean conversion ratios were achieved

at higher stocking densities, $16 > 8 > 4$ worms. During subsequent 2-week feeding periods (weeks 4, 6 and 8) differences in cumulative conversion between stocking densities decreased. Higher mean conversion ratios were achieved for large juveniles, initially (2 weeks), although, mean cumulative conversion ratios after 8 weeks were broadly similar for small and large juveniles at high stocking densities (8 and 16 worms). Small juveniles produced a significantly lower final cumulative conversion (8 weeks) than large juveniles at a density of 4 worms ($p < 0.05$, Student t-test with Welch's correction).

Figure 15 – Total worm production as % processed-WPS; large juveniles reared at stocking densities of:
16: ●—, 8: *—, 4: X—

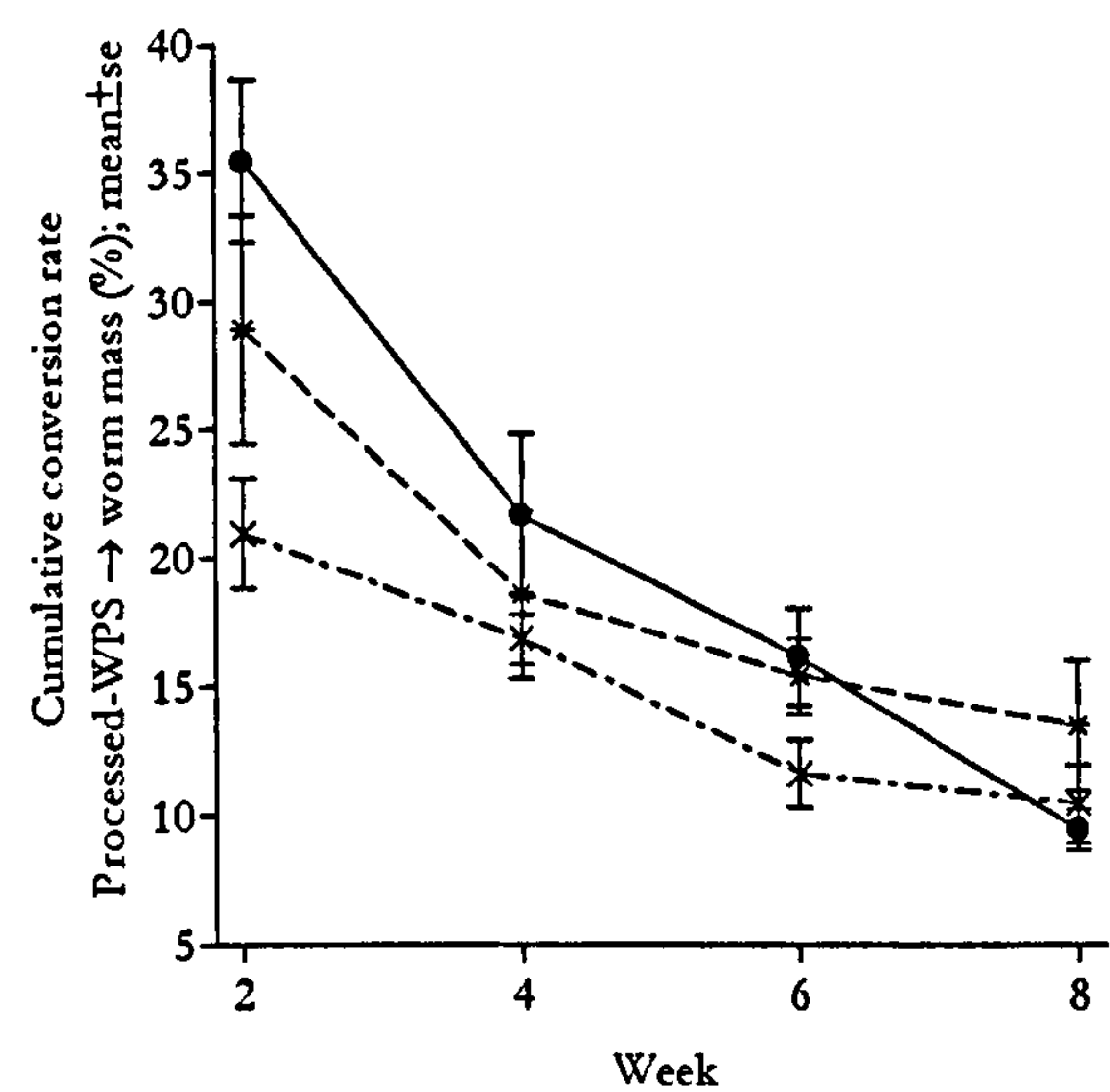


Table 13 – Cumulative conversion of processed-WPS into worm mass (%); mean ± se

Week	Stocking density			
	ANOVA	16	8	4
2	*	35.6±3.2 ^a	28.8±4.4 ^{ab}	21.0±2.1 ^b
4	NS	21.6±3.1 ^a	18.4±3.2 ^a	16.8±1.0 ^a
6	NS	16.2±1.9 ^a	15.2±1.4 ^a	11.6±1.2 ^a
8	NS	9.4±0.7 ^a	13.8±2.6 ^a	10.4±1.4 ^a

NS : Not significant ;One-way ANOVA, $p > 0.05$

Figure 15 and Table 13 show the conversion of WPS to worm mass in terms of actual quantities of processed-WPS (casts) produced by large juveniles. Again, higher conversion ratios were achieved at higher worm stocking densities initially ($p < 0.05$), but at subsequent feeding periods cumulative conversion diminished, as did differences between stocking densities.

3.3.4 Reproduction

Cocoon production

Figure 16 – Total production of cocoons;
small juveniles reared at stocking densities of:
16: ●—, 8: *—, 4: X—, 2: +—

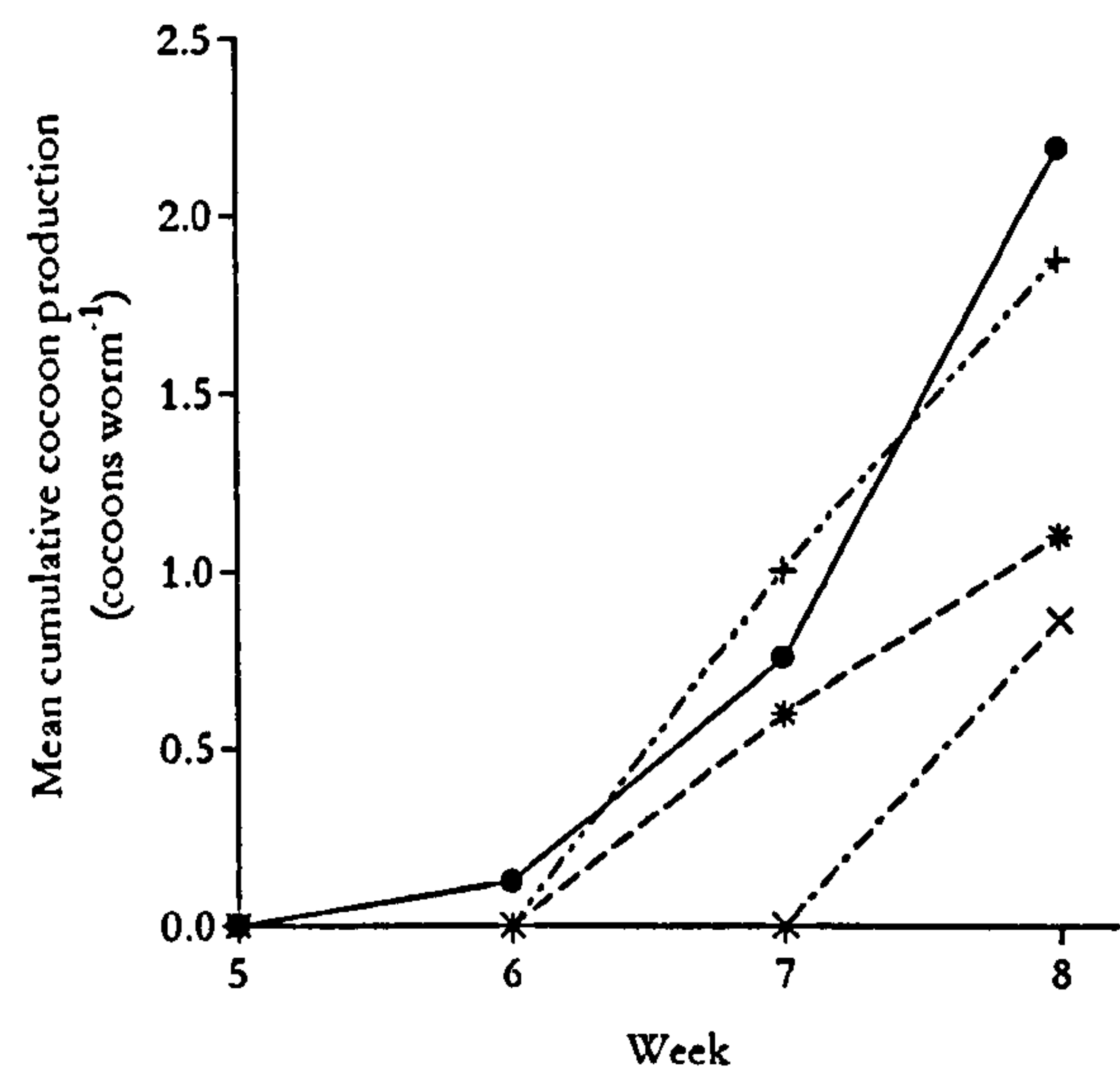
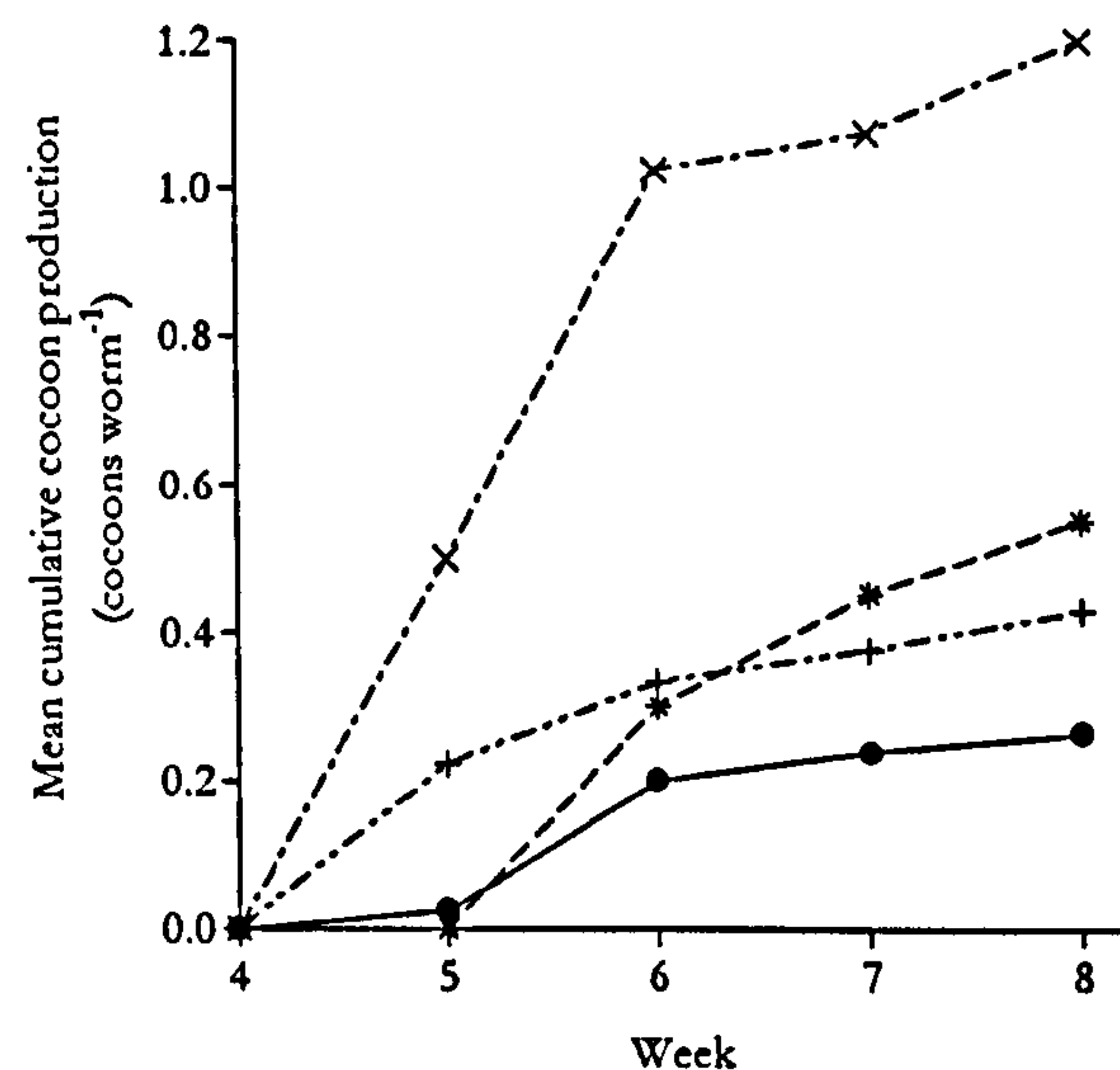


Figure 17 – Total production of cocoons;
large juveniles reared at stocking densities of:
16: ●—, 8: *—, 4: X—, 2: +—



The onset of cocoon production was observed at all densities for surviving cultures of small and large juveniles during the 8-week experimental period (Figure 16 and Figure 17). The onset of cocoon production was observed between 5 to 7 weeks for small juveniles, and 4 to 5 weeks for large juveniles.

In surviving cultures of small juveniles, stocking densities of 2 and 16 worms produced higher mean cumulative cocoon production per worm over 8 weeks (1.88 and 2.19, respectively) than 4 and 8 worms (0.86 and 1.09, respectively).

However, in cultures of large juveniles, stocking densities of 4 and 8 worms produced higher mean cumulative cocoon production per worm over 8 weeks (1.20 and 0.55, respectively) than 2 and 16 worms (0.43 and 0.27, respectively). Cumulative cocoon production of large juveniles appeared to begin to reach a plateau (ceasation) after week 6 at all stocking densities.

Table 14 – Total cocoon production after 8 weeks

Stocking density	Small juveniles			Large juveniles		
	Range	Replicates without cocoons	Mean±se	Range	Replicates without cocoons	Mean±se
2	4-11 (N=4)	50%	3.8±2.6	1-3 (N=5)	60%	0.9±0.3
4	1-13 (N=7)	29%	3.4±1.7	1-15 (N=10)	20%	4.8±1.6
8	4-15 (N=4)	0%	8.8±2.3	1-11 (N=5)	0%	4.4±1.8
16	35 (N=1)	0%	35	2-8 (N=4)	0%	4.3±1.4

N = number of surviving replicates

Total cocoon production per surviving replicate-pot was found to be very variable (Table 14). Many surviving replicates did not produce cocoons at lower stocking densities (50-60% - 2

worms, 20-29% - 4 worms). High levels of mortality (section 3.3.5) greatly reduced the number of surviving replicates, making results difficult to interpret (*e.g.*, N=1 for the stocking culture of 16 small juveniles). Due to large levels of variance, statistical analysis was ineffective.

Cocoon viability (hatching rates)

Figure 18 – Hatching rate of cocoons ('survival' curves); collected at weeks:

5: ○----; 6: □ - - -; 7: ● - - , 8: ■—

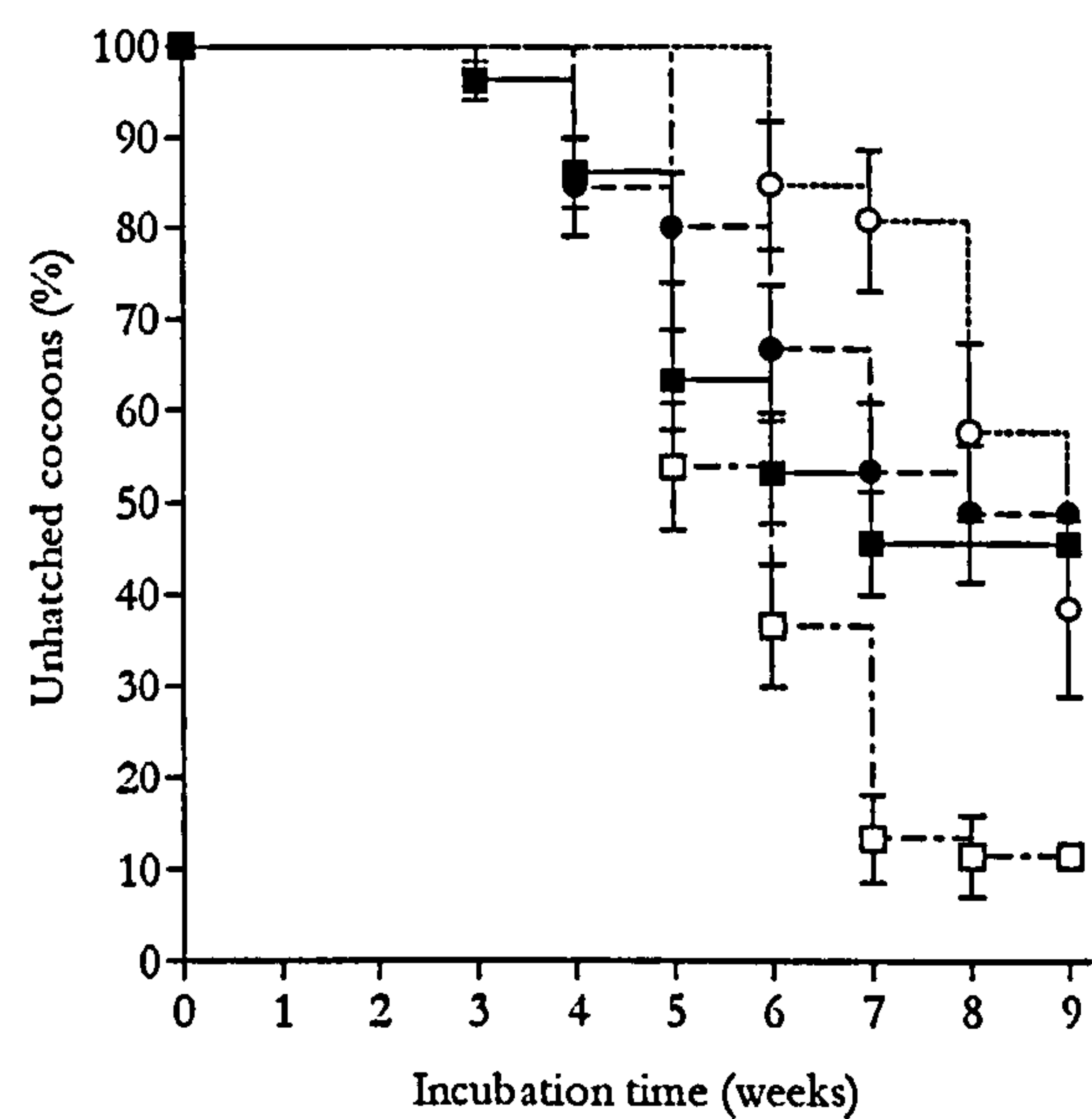


Table 15 – ‘Survival’ curve analysis; mean±se

	Week of cocoon collection			
	5	6	7	8
Log-rank tests	A	b	a	a
Median hatching week	9	6	8	7
Final % hatched	61.5%	88.5%	51.1%	54.4%
Log-rank test for trend	NS; χ^2 0.93; 1 df			

NS : Not significant, $p>0.05$

Cocoons began to hatch at between 3 and 6 weeks (Figure 18). Cocoons collected at weeks 6 hatched at a significantly higher rate than cocoons collected at all other weeks ($p<0.05$, Log-rank tests, Table 15). Median hatching was achieved after 6 weeks as compared to 7, 8 and 9 weeks. The final proportion of cocoons that hatched during the experimental period were higher for cocoons collected at weeks 5 and 6 - 88.5% and 61.5%, respectively. Whereas, the final proportion of cocoons which hatched from cocoons collected at weeks 7 and 8 were 51.1% and 54.4%, respectively.

Polyembryony

Table 16 – Number of hatchlings per cocoon

Week of Cocoon collection	Number of cocoons hatched	Number of Hatchlings	Polyembryony hatchlings cocoon ⁻¹
5	14	14	1.00
6	46	48	1.04
7	19	19	1.00
8	37	40	1.08
Mean±se			1.03±0.02

All cocoons showed very low levels of polyembryony, most cocoons producing one hatching. In instances of polyembryony a maximum of two hatchlings per cocoon were produced (cocoons collected at weeks 6 and 8).

3.3.5 Earthworm mortality

Figure 19 – Survival curve of earthworms; small juveniles reared at stocking densities of: 16: ●—; 8: *—; 4: X—; 2: +—; 1: ○—

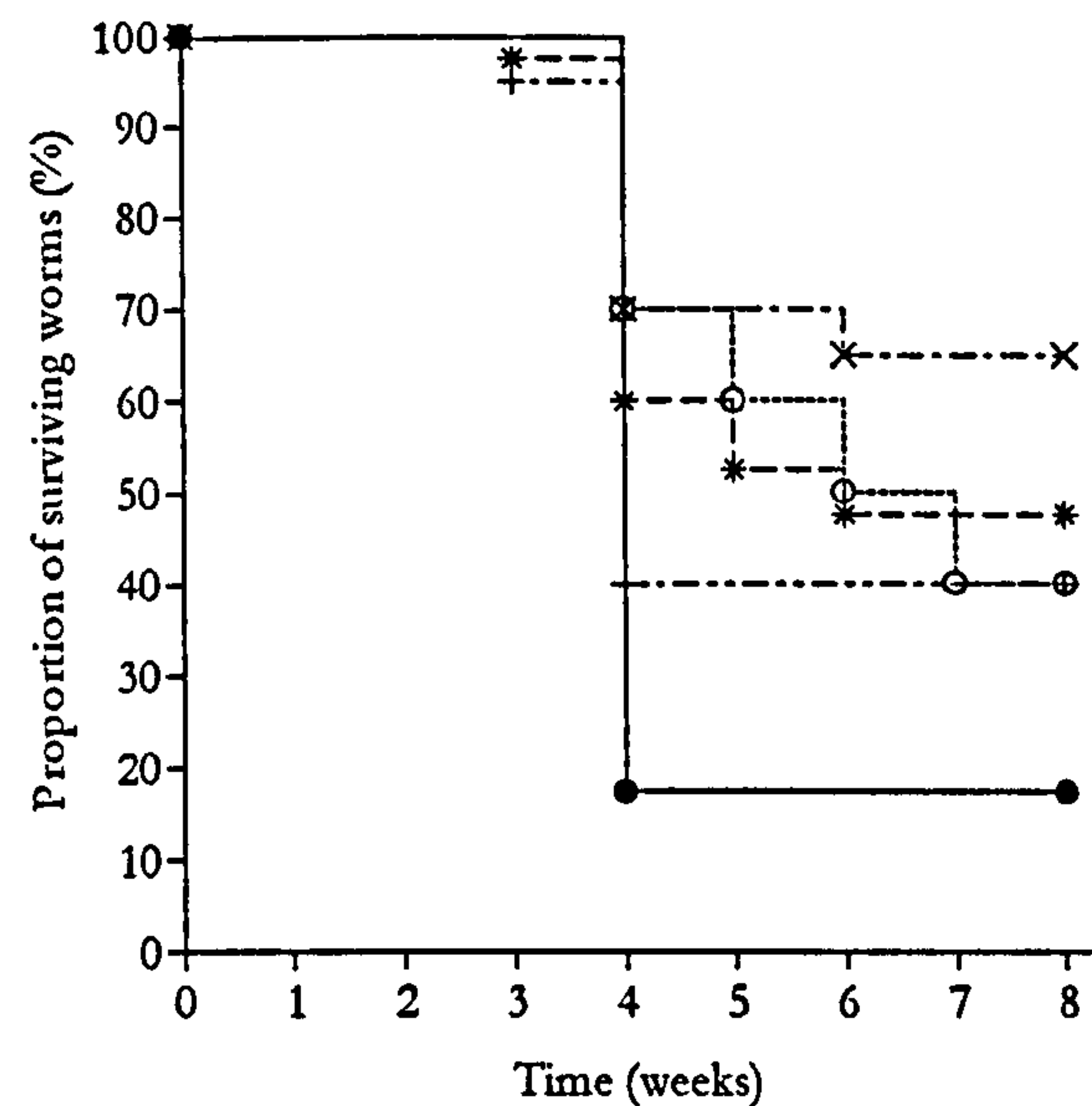


Figure 20 – Survival curve of earthworms; large juveniles reared at stocking densities of: 16: ●—; 8: *—; 4: X—; 2: +—

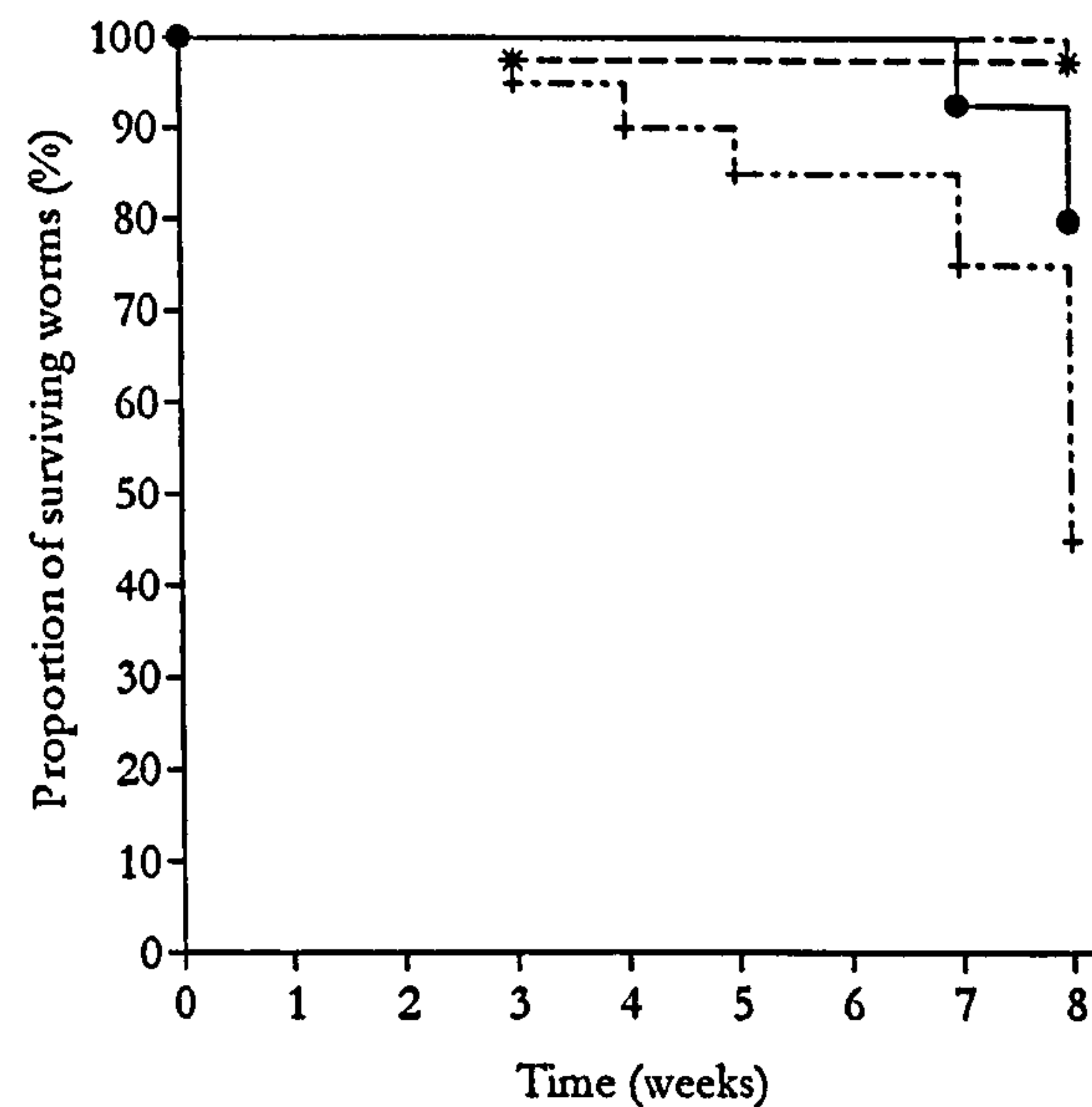


Table 17 – Survival curve analysis

Small juveniles					Large juveniles				
	Stocking density					Stocking density			
	16	8	4	2		16	8	4	2
Log-rank tests	a	b	b b	a b c	Log-rank tests	a	b	b	c
Median survival week	4	6	U	4	Median survival week	U	U	U	8
Final survival	17.5%	47.5%	65.0%	40.0%	Final survival	80.0%	97.5%	95.0%	45.0%
Log-rank test for trend	p<0.05; χ^2 5.92; 1 df				Log-rank test for trend	p>0.05; χ^2 1.95; 1 df			

abc : Different letters indicate significant differences (p<0.05, Log-rank tests); 1st letter difference from 1st column, 2nd letter difference from 2nd column etc.
U : undefined

Higher levels of worm survival were observed for large juveniles during the 8-week experimental period (cf. Figure 19 with Figure 20). High levels of survival were observed for large juveniles at densities 16, 8 and 4 worms (80, 97.5 and 95%, respectively), whereas final levels of survival were much lower for small juveniles employing similar worm numbers (17.5, 47.5 and 65%, respectively).

The week at which median survival (50%) was reached for large juveniles was undefined (U) for densities of 16, 8, and 4 worms, as survival exceeded 50% over the 8-week experimental period. Only at a density of 2 worms did survival descend below 50%. However, median survival was reached within 4 to 6 weeks at densities of 16, 8 and 2 worms for small juveniles and only at a density of 4 worms did survival remain above 50%.

A significant trend between stocking density and survival was observed for small juveniles ($p < 0.05$, Log-rank test for trend - Table 17), whereas no significant trend was observed for large juveniles ($p > 0.05$, Log-rank test for trend). Decreased stocking density resulted in significantly increased worm survival for both small and large juveniles, cf. 16 worms with 8 and 4 worms ($p < 0.05$, Log-rank tests). However, at a stocking density of 2 worms, survival was significantly lower than for 4 or 8 large juveniles ($p < 0.05$, Log-rank tests).

3.3.6 Substrate processing

Figure 21 – WPS processed per unit of worm mass; large juveniles reared at stocking densities of: 16: ●—, 8: *—, 4: X---

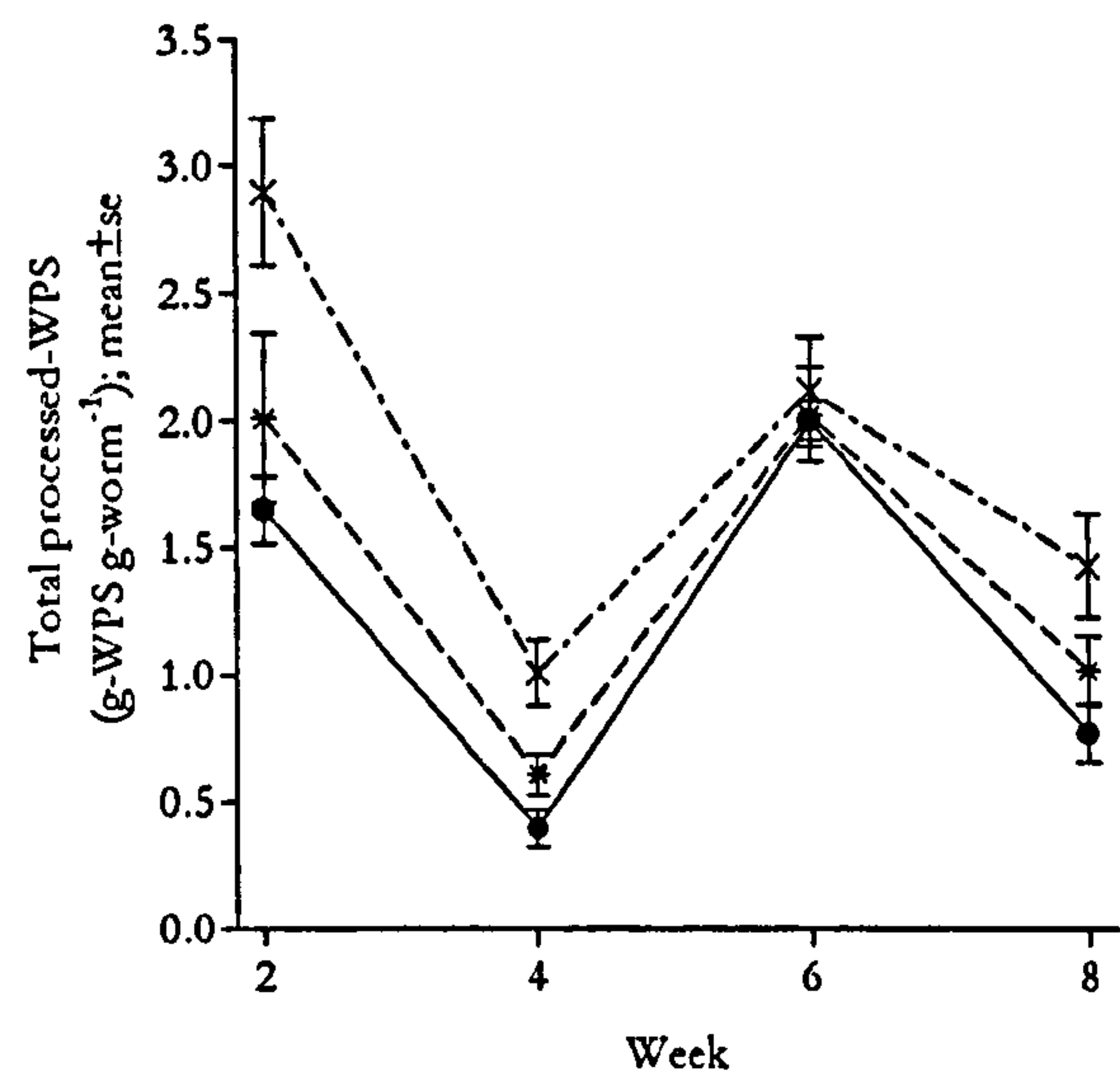


Table 18 – Changing processing rate of WPS by large juveniles; mean±se

Week	Stocking density			
	ANOVA	16	8	4
2	*	1.65±0.13 ^a	2.01±0.33 ^{ab}	2.91±0.29 ^b
4	**	0.40±0.07 ^a	0.60±0.07 ^a	1.01±0.13 ^b
6	NS	1.99±0.07 ^a	2.02±0.19 ^a	2.11±0.21 ^a
8	*	0.77±0.11 ^a	1.03±0.13 ^{ab}	1.44±0.20 ^b

One-way ANOVA: NS (not significant) $p>0.05$; * $p<0.05$; ** $p<0.01$; *** $p<0.001$;
abc : Different letters indicate significant differences ($p<0.05$, Tukey-Kramer multiple comparisons test)

Figure 22 – Total processed-WPS; large juveniles reared at stocking densities of: 16: ●—, 8: *—, 4: X---

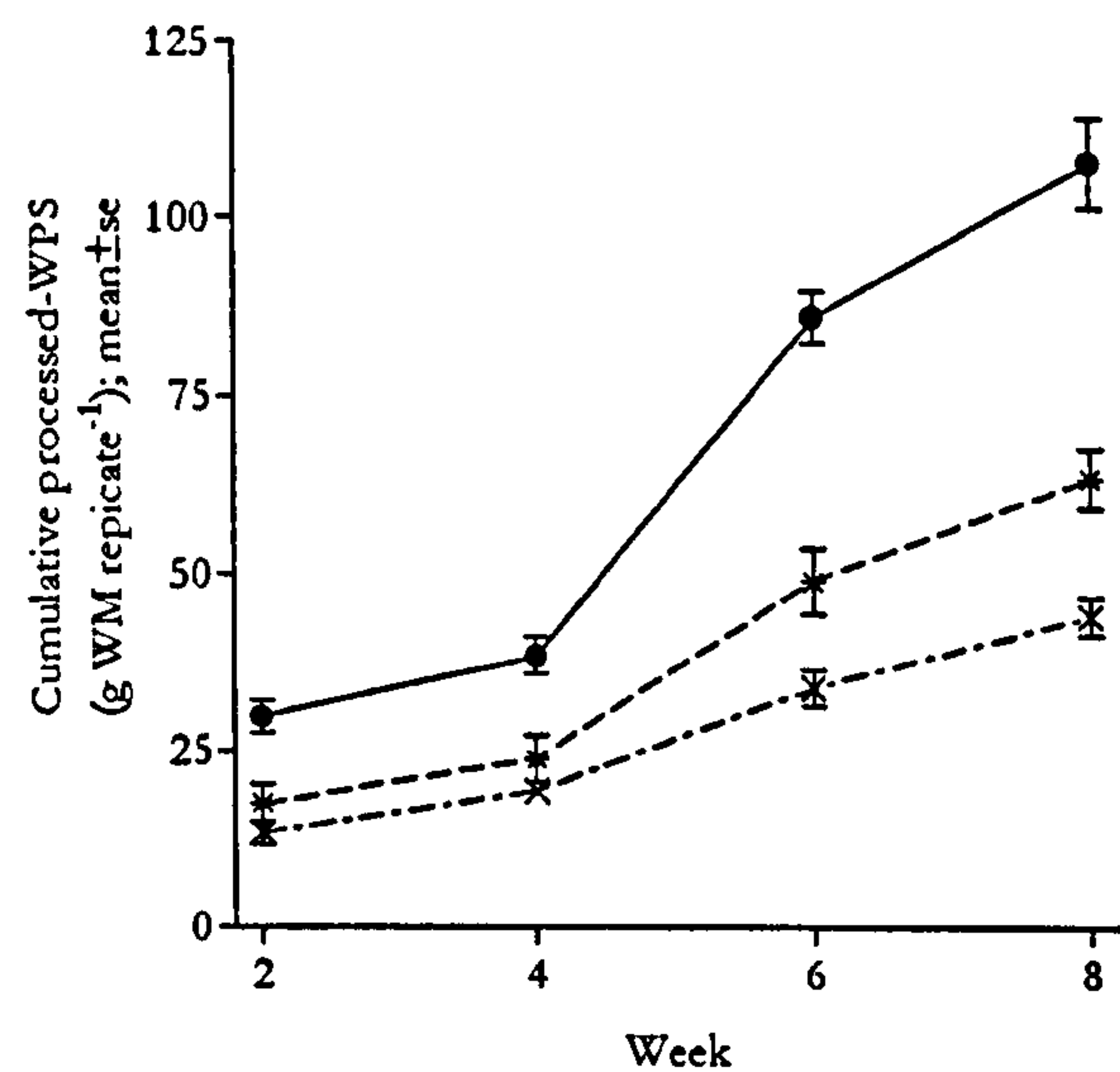


Table 19 – Cumulative processing of WPS by large juveniles; mean±se

Week	Stocking density			
	ANOVA	16	8	4
2	***	29.8±2.3 ^a	17.5±2.7 ^b	13.3±1.7 ^b
4	***	38.4±2.5 ^a	23.9±3.2 ^b	19.3±1.2 ^b
6	***	86.1±3.5 ^a	49.0±4.6 ^b	33.9±2.6 ^c
8	***	108.0±6.4 ^a	63.5±4.2 ^b	44.0±2.7 ^c

The mass of WPS processed in 2 weeks, per mass of worm, decreased with increased stocking density (Figure 21 and Table 18), although processing rates also appeared to change between each 2-week feeding period. This was supported by two-way ANOVA results, which showed both density and 2-week feeding period had a significant effect on processing rate ($p<0.0001$); feeding period accounted for 63.3%, whereas density only accounted for 11.7% of the total variation. No significant interaction between feeding period and density was observed ($p>0.05$).

Despite reduced processing rates (in proportion to worm mass) at the highest stocking density (16 worms), the total quantity of WPS processed (Figure 22 and Table 19) was significantly greater ($p<0.05$, Tukey-Kramer multiple comparisons test). The total quantity of WPS processed by 16 large juveniles, at the end of the 8-week experimental period, was equivalent to 27% of the total WPS-feed added; 16% - 8 worms; and 11% - 4 worms.

Total organic matter (TOM) content

Table 20 – TOM content of unprocessed and processed WPS (casts) %DM; mean±se.

Week	Ingested WPS (casts)				Non-ingested WPS			
	Density			Row mean±se	Density			Row mean±se
	4	8	16		4	8	16	
2	73.3	76.5	75.2	75.0±0.9	81.0	80.4	79.9	80.4±0.3
4	74.3	72.0	73.2	73.2±0.7	78.7	78.6	78.4	78.6±0.1
6	79.9	79.1	78.7	79.2±0.4	82.4	81.5	81.5	81.8±0.3
8	74.1	73.5	73.0	73.5±0.3	78.1	77.8	77.5	77.8±0.2
Column mean±se	75.4±1.5 ^a	75.3±1.6 ^a	75.0±1.3 ^a	75.2±0.8*	80.1±1.0 ^a	79.6±0.9 ^b	79.3±0.9 ^b	79.7±0.5*
TOM loss %ash	45.3%	45.6%	46.4%	45.8%	28.4%	30.4%	31.5%	30.1%

* italic figures: means for all organic matter values

abc : Different letters indicate significant differences (p<0.05, Tukey–Kramer multiple comparisons test; values between columns were paired)

TOM contents of both ingested and non-ingested WPS were significantly (p<0.01, Student t-test) below organic matter content of fresh WPS, 84.9±0.5%DM. Casts showed a 9.7%DM, and non-ingested WPS a 5.2%DM, decrease in TOM. These were equivalent to TOM losses of 45.8%ash and 30.1%ash, respectively. Therefore, ingestion by earthworms increased mean TOM losses by 52%, during each 2-week period.

The mean TOM content of WPS casts, 75.2%DM, was significantly below that for non-ingested WPS, 79.7%DM (p<0.0001, paired Student t-test).

Mean TOM contents of both casts and unprocessed WPS produced at different densities (column means, Table 20) appeared to show a systematic decrease in TOM contents with increasing stocking density for both casts and non-ingested WPS.

Stocking density had no significant effect upon the TOM content of casts (P>0.05, two-way ANOVA); whereas significant variations between 2-week processing periods were observed (p < 0.01, F = 15.7, accounting for 88.4% of the total variance).

However, stocking density did have a significant effect upon the TOM content of non-ingested WPS (p < 0.01, F = 11.3, two-way ANOVA), although this only accounted for 3.5% of the total variance. Each 2-week processing period, again, had a greater effect (p < 0.0001, F = 204.6, two-way ANOVA), accounting for 95.5% of the total variance.

3.3.7 Aluminium content of final worm cultures

Table 21 – Aluminium content of worms after culture on WPS for 8 weeks

Stocking culture	Aluminium concentration in earthworm biomass	Total worm mass (g)					
	After (8 weeks)	Before		After 8 weeks		Increase (g)	
	mg kg ⁻¹	WM	DM	WM	DM	WM	DM
Large juveniles	252	125	23.8	267	50.7	142	26.9
Small juveniles	422	12.6	2.40	93.4	17.7	80.8	15.3

The aluminium content of surviving worms derived from stocking cultures of large juveniles (252 mg kg⁻¹) was 67.5% (170 mg kg⁻¹) lower than surviving worms derived from cultures of small juveniles (422 mg kg⁻¹). Total levels of growth during culture in WPS were 26.9 mg DM and 15.3 mg DM for large and small juveniles, respectively. This equated to 53.0% and 86.5% of their final worm mass, respectively. The difference in the proportion of total worm mass obtained during growth on WPS feed was calculated as 63.2%, similar to the 67.5% difference in final aluminium concentrations of worms derived from cultures of small and large juveniles. As analyses were determined for composite samples of pooled worm cultures, replicate data were not produced, making correlations impossible.

3.4 Discussion

WPS nutrition

WPS feedstock moisture contents of 79.4% (batch A) and 74.0% (batch B) fell within the optimal range of 73-80% for the growth and reproduction of *D. veneta* cultured at 15°C when grown on cattle manure (Muyima *et al.* 1994). Viljoen *et al.* (1992) maintained similar moisture contents, 75-80%, during the investigation of the influence of temperature upon *D. veneta*.

These moisture contents were also broadly similar to paper sludges employed by Fayolle *et al.* (1997) for the culture of *D. veneta* - 78.3%; Elvira *et al.* (1996 & 1997) for the culture of *Eisenia andrei* - 81.1%; and Butt (1993) for the culture of *Lumbricus terrestris* - 80.4%.

The total organic matter and carbon contents of the WPS feedstock (84.4-85.3%DM and 46.4-46.9%DM, respectively) were also comparable to other WPS used for earthworm culture: 89.9 and 45.9%DM (Fayolle *et al.* 1997); 89.8 and 46.7%DM (Butt 1993).

Organic matter was present mainly in the form of cellulose fibres (34%), again comparable to WPS used for worm culture, 49.1% (*op. cit.* 1993), and other studies, 31.6% (Gijzen *et al.* 1990), 23.4% (Kannan *et al.* 1990).

The C:N ratio of the WPS feedstock (16.9-22.2:1) was much lower than others, *cf.* 45.4:1 (*op. cit.* 1997) and 93:1 (*op. cit.* 1993), due to higher total nitrogen contents (2.11-2.75%DM), *cf.* 1.0%DM (*op. cit.* 1997) and 0.5%DM (*op. cit.* 1993). Neuhauser *et al.* (1980b) suggested a C:N ratio of between 15-35:1 for the culture of *Eisenia foetida*, but there is little other published data on appropriate C:N ratios for other earthworm species.

Total water extractable nutrient contents (N, P, K, Mg and Ca) of the WPS feedstock amounted to between 0.24% (batch B) to 0.30%WM (batch A), below an upper limit of 0.5% suggested for *E. fetida* culture (Edwards 1988). An electrical conductivity (EC) of 984 (batch B) $\mu\text{S cm}^{-1}$ and 1360 $\mu\text{S cm}^{-1}$ (batch A) was below a range of 1500 to 3000 $\mu\text{S cm}^{-1}$ found not to be harmful to similar earthworm species (*E. fetida*), when cultured on horse manure (Hartenstein *et al.* 1979b).

The WPS feedstock used in this study was more acidic (pH 6.0) than WPS used in other studies: 7.1 (*op. cit.* 1993), 9.2 (Elvira *et al.* 1996 & 1997); but within the pH range of 5 to 9 recommended for *E. fetida* (Edwards 1988).

Worm mass production

The total production of *D. veneta* mass was significantly affected by stocking density using both small and large juveniles. Higher stocking densities produced higher final total worm mass, following significant trends ($p < 0.0001$, ANOVA). A similar finding was observed when

culturing *Eisenia fetida* at densities of 4, 8, 12 and 16 worms per 0.3 litres of horse-manure or activated sludge (Hartenstein *et al.* 1979b). Edwards (1983) also found increasing total worm mass production with increasing stocking density; culturing *E. fetida* on 40 g WM of potato solids in 0.125 litre containers at densities of 1, 2, 4, 8, and 16 worms.

Higher total worm mass production, at higher worm densities, can be attributed to the fact that increased stocking density did not proportionately diminish final individual worm mass and corresponding growth rate (*i.e.* a doubling in stocking density reduced biomass production by <50%). Although, individual worm growth was decreased by increased stocking density, *q.f.* 12.1-13.7 mg worm⁻¹ d⁻¹ (16 worms) with 21.4-22.3 mg worm⁻¹ d⁻¹ (2 worms), statistical differences were not observed due partly to large levels of variance within the data. High levels of variation can reduce statistical 'power', increasing the chance of Type II statistical errors (incorrect acceptance of the 'null hypothesis'), suggesting further experimental replication, or 'power-analysis' may be required to resolve statistical differences (*q.v.* Motulsky 1995).

Lower final levels of total worm biomass at lower stocking densities (2-4) suggested that, in terms of worm mass production, these densities were inefficient. However, significantly higher final individual worm masses obtained for large juveniles ($p < 0.05$, t-tests) suggested this was not due to worms reaching maximum obtainable mass. It appeared that some factor, other than stocking density, restricted final individual worm mass.

Non-significant differences between final individual worm mass and corresponding growth rates were also attributed to the feeding regime employed (100g WPS 2-weeks⁻¹). Decreases in individual worm growth, due to intra-specific competition, are caused by competition for food (*e.g.*, *E. fetida*: Hartenstein *et al.* 1979a & b; Neuhauser *et al.* 1980a) and cast toxicity (Kaplan *et al.* 1980). During this investigation the removal and replenishment of food alleviated these effects somewhat. In studies which have shown more significant effects of stocking density upon earthworm growth, a fixed quantity of feed was often employed (*e.g.*, Frederickson *et al.* 1997; Dominguez & Edwards 1997).

Highest levels of total worm biomass, predicted by non-linear regression, were achieved at densities of 16 juveniles, 11.6-11.9 g-worm per replicate-pot after 8 weeks. This was equivalent to a final density of 11.6-11.9g per 157-189 ml of WPS = 61-76 g-*D. veneta* l⁻¹ WPS 2-weeks⁻¹. This was comparable to a maximum of 75 g-*D. veneta* l⁻¹ pot volume after 28 weeks, without a reproductive increase in worm numbers, calculated from the findings of Fayolle *et al.* (1997); and 80 g-*E. andrei* l⁻¹ with a reproductive increase in worm numbers, cultured upon an excess feed of pig-manure (Reeh 1992). Hartenstein (1981) obtained a maximum carrying capacity of approximately 86g-*E. fetida* l⁻¹ horse manure (-9.5g per 110 ml), at a density of 22 worms, over 7 weeks.

Final individual worm mass increases, e.g. 1219-1290 mg worm⁻¹ for a stocking density of 2 worms after 8 weeks, were lower than observed in other studies. Fayolle *et al.* (1997) achieved an increase of 1680 mg worm⁻¹ after 8 weeks, at a higher stocking density (5 *D. veneta* hatchlings pot⁻¹), at 20°C, WPS supplied at a rate of 230 mg WM g-worm⁻¹ d⁻¹ (replaced every 10-11 days). However, during their study (*op. cit.* 1997), pot-size was increased from 0.5 l to 0.9, after 6 weeks.

An increase of 793-1132 mg worm⁻¹ after 8 weeks, observed for a stocking rate of 4 juveniles, observed during this study, was more comparable to a 750 mg worm⁻¹ increase observed for 5 worms grown on horse-manure and peat at 20°C (*op. cit.* 1997).

The lowest individual worm mass increases (671-765 mg worm⁻¹), observed during this study, for a stocking density of 16 worms, showed WPS compared favourably to horse-manure used as a feed for *D. veneta* (*op. cit.* 1997).

Large juveniles produced significantly higher final masses (1492-2091 mg) than small-juveniles (888-1383 mg) at all densities ($p < 0.05$, Tukey-Kramer test). Highest final individual worm masses (2091 and 1933 mg) were achieved for large-juveniles at stocking densities of 2 and 4 worms, respectively.

These values were comparable to a highest mean individual biomass of 2350 mg, obtained for *D. veneta* cultured on cattle-manure (replaced every 5 days) for 28 weeks (Viljoen *et al.* 1992); stocking density was not reported. However, Fayolle *et al.* (1997) obtained a larger maximum individual biomass of 2700 mg after 18 weeks culture on WPS. Again, this suggested that final individual worm mass was restricted by other factors in addition to stocking density.

Individual worm growth rates of 21.4-22.3 mg worm⁻¹ day⁻¹ at a stocking density of 2 worms were lower than 30 mg worm⁻¹ d⁻¹, observed for *D. veneta* over 8 weeks at 20°C (Fayolle *et al.* 1997). Lowest individual worm growth rates of 12.1-13.7 at a stocking density of 16 worms were comparable to 11.9 mg worm⁻¹ d⁻¹ recorded for *D. veneta* cultured on cattle-manure over 28 weeks (Viljoen *et al.* 1992).

Higher total worm mass production with increased stocking density, with the production of individually smaller worms, observed during this study, has also been observed by others. Dominguez & Edwards (1997) observed a larger total worm mass production per unit of waste with increased worm density, although individual worm (*E. fetida*) mass was reduced. Reeh (1992) found that while stocking density had an effect on worm (*D. Veneta*) numbers, total biomass was little affected over 33 weeks culture period using a substrate of pig manure; a common ecological phenomenon (Begon *et al.* 1986). For example, the 'law of constant yield' shown amongst crop plants, *i.e.* plants planted above certain densities will show no

increase in overall yield due to increasing intraspecific competition between plants for nutrients, space and light (The Open University 1996b).

Conversion ratios

Higher total worm mass production at higher stocking densities and using larger juvenile worms, translated to higher levels of the conversion of WPS feed added into worm mass ($p < 0.05$, ANOVA). Highest conversion ratios of 5.1% (small-juveniles) and 10.3% (large-juveniles) for the first 2-week feeding period at a stocking density of 16 worms, diminished to 3.0 and 2.4%, respectively, by week 8 (fourth feeding-period). A stocking density of 4 worms produced a conversion of 1.5% (small-juveniles) and 2.7% (large-juveniles) after 2 weeks, diminishing to 0.8 and 1.13%, respectively, by week 8.

Maximum conversion ratios of 4.5 to 4.9% were observed for 1-16 *E. fetida* cultured on 40g potato waste (Edwards 1983); and 2.8 to 3.0% for 4 *E. fetida* cultured on 60 g undigested cattle-solids, after 6 weeks (Frederickson & Knight 1988). Though, higher conversion ratios have been observed for other earthworm species/wastes (*op. cit.* 1983 and 1988), a fixed quantity of feed was used, whereas in this study an excess of WPS feed was utilised.

Conversion ratios based upon actual quantities of WPS ingested by worms were much higher, and revealed that higher conversion ratios were obtained with higher stocking densities after 2 weeks (*cf.* 35.6% - 16 worms with 21% - 4 worms). However, final conversion ratios at different stocking densities were similar after 8 weeks (*cf.* $9.4 \pm 0.7\%$ - 16 worms with $10.4 \pm 1.4\%$ - 4 worms). Reductions in conversion ratios (based on feed ingested) over time have also been observed for *E. andrei*, cultured on pig manure in excess of nutritional requirements, with maximum conversion ratios of 6 to 10% (Reeh 1992). It is impossible to say whether the higher conversion ratios observed during this study were due to the earthworm species (*D. veneta*), stocking density, or the nutritional value of the WPS feedstock employed, and this requires further investigation.

Worm reproduction – cocoon production and hatching rates

As expected cultures of large juveniles began to produce cocoons earlier than small juveniles. Cocoon production commenced at between 5-7 weeks for stocking cultures of small juveniles and 4-5 weeks for stocking cultures of large juveniles. This compares to 8 weeks for *D. veneta* cultured on sewage-sludge (Neuhauser *et al.* 1988); 8.6 weeks for day-old hatchlings of *D. veneta* cultured on cattle-manure at 25°C (Viljoen *et al.* 1992); 6-9 weeks for 'newly-hatched' *D. veneta* cultured on paper-sludge at 20-25°C (Fayolle *et al.* 1997). The earlier onset of reproduction observed during this study was attributed to the greater initial age of the stocking cultures employed.

Cocoon production rates were generally very low, falling from 0.27 cocoons worm⁻¹ (16 large juveniles) to 1.88 cocoons worm⁻¹ (2 small juveniles). A total cocoon production of 2.19 cocoons worm⁻¹ for one stocking culture of 16 small juveniles suggested higher levels of reproduction were possible. However, this figure was obtained from only one replicate, which may have been atypical.

Maximum cocoon production was observed for cultures derived from 4 large juveniles after week 6 (0.51 cocoons worm⁻¹ wk⁻¹) and 2 small juveniles after week 8 (0.94 cocoons worm⁻¹ wk⁻¹). These rates were lower than observed during other investigations: 2.0 cocoons worm⁻¹ wk⁻¹ (Lofs-Holmin 1985); 1.6 (Edwards 1988); 5 (Neuhauser *et al.* 1988); 1.96 at 25°C, 1.19 at 20°C (Viljoen *et al.* 1992); and 4.5 at 20°C on paper-sludge [mistakenly reported as cocoons worm⁻¹ d⁻¹] (Fayolle *et al.* 1997).

The experimental period used in this study was much shorter than other studies (20 weeks, *op. cit.* 1988; 28 weeks *op. cit.* 1992; 27 weeks *op. cit.* 1997), and may have not allowed full production rates to be reached. Viljoen *et al.* (1992) did not observe peak cocoon production per worm until after 14 weeks culture.

However, cocoon production of worms derived from large juveniles appeared to cease 1-2 weeks after its onset, suggesting the onset of some form of inhibition effect during culture.

Generally, higher stocking densities produced fewer cocoons per worm, although no systematic trend was observed. Similar effects of density upon cocoon production have been observed for *E. fetida* (Hartenstein *et al.* 1979a) and *E. andrei* (Reeh 1992).

Levels of total cocoon production were erratic, possibly related to mortality rates (see below). Some replicates showed high levels of cocoon production, while others showed none. This produced very large levels of variance, making statistical analysis ineffective.

Times at which cocoons began to hatch ranged between 3 and 6 weeks incubation, some cocoons taking as long as 10 weeks to hatch. This is comparable to incubation times of 5 and 6 weeks (50% hatching) observed for *D. veneta* cocoons on moist peat and plaster (Fayolle *et al.* 1997); a mean of 6 weeks at 25°C (Viljoen *et al.* 1992), and 6 to 18 weeks reported by Edwards (1988).

Cocoons collected at week 6 showed lower median (50%) hatching time (6 weeks incubation) compared to those collected at all other weeks (7-9 weeks incubation) ($p < 0.05$, Log-rank tests).

Cocoon viability (% hatched) varied between 51 and 89%. Cocoons collected after 7 and 8 weeks showed lower viability (51% and 54%, respectively) compared to those collected at weeks 5 and 6 (62% and 89%, respectively).

Cocoon viability compared favourably to a maximum of 37.8% observed by Viljoen *et al.* (1992), but unfavourably to a minimum of 84% observed by Fayolle *et al.* (1997), and means of 81.2% (Edwards 1988), 78% Neuhauser *et al.* (1988).

These studies (*op. cit.* 1992, 1997 and 1988) demonstrate the high variability of cocoon viability, which may be caused by different culturing conditions.

Polyembryony was typically low, with a mean of 1.03 ± 0.02 worms cocoon⁻¹, as seen in other investigations (1.1 worm cocoon⁻¹, Edwards 1988; 1.1 at 25°C and 1.2 at 15°C, Viljoen *et al.* 1992; 1.02, Fayolle *et al.* 1997).

Worm survival

Survival analysis revealed increased levels of mortality with increased stocking density, in cultures derived from small juveniles ($p < 0.05$, Log-rank test for trend). All but a density of 4 worms (total mortality – 35%) showed 50% mortality within 4 to 6 weeks. A culture of 16 worms showed the highest total mortality, 82.5% (4 out of 5 replicate pots showed 100% mortality); 8 worms – 52.5%; 2 worms – 60%.

Such high levels of mortality were comparable to levels (<50 to >75%) observed by Elvira *et al.* (1997) when utilising *Eisenia andrei* to process paper-sludge amended with sewage sludge, pig slurry and poultry slurry. Fayolle *et al.* (1997) observed a much lower mortality ($\leq 6\%$) of *D. veneta*, utilising paper sludge as the sole food source.

Cultures derived from large juveniles showed much lower levels of mortality, stocking densities of 16, 8, 4 and 2 worms showing 20, 2.5, 5, and 55% mortality, respectively. Interestingly, densities of 16 and 2 worms gave the highest levels of mortality, as observed for small juveniles. This suggested that other factors, as well as high stocking density, were possibly attributing to earthworm mortality.

WPS processing

As expected, higher levels of total WPS processing were observed using higher stocking densities ($p < 0.0001$, two-way ANOVA), *e.g.*, 27% for 16 worms and 11% for 4 worms, after 8 weeks. However, the mass of processed WPS per unit mass of worm was significantly lower ($p < 0.0001$, two-way ANOVA), *e.g.*, 1.65 g g-worm⁻¹ for 16 worms and 2.91 g g-worm⁻¹ for 4 worms, after 2 weeks. Similar phenomena were observed by Reeh (1992), when culturing different densities (3, 6 and 12 worms) of *E. andrei* on pig manure.

A maximum ingestion rate of 1.5 g-WPS g-worm⁻¹ week⁻¹ (0.21 g g-worm day⁻¹) was observed for the first 2-week feeding period at a stocking density of 4 worms. This rate was low compared to other studies: 2-volumes volume-worm⁻¹ d⁻¹ *Eisenia fetida* (Hartenstein &

Hartenstein 1981); 0.5-1.0 g g-worm⁻¹ d⁻¹ for *Lumbricus rubellus* (Pierce 1978); 0.3-0.5 g g-worm⁻¹ d⁻¹ for *D. veneta* (Lofs-Holmin 1985); and a maximum of 0.8 g g-worm⁻¹ d⁻¹ for 6 *D. veneta* per 1 l pot (Reeh 1992). However, these processing rates also compare favourably to other trials: 0.13 g g-worm⁻¹ d⁻¹ (*E. fetida*) using a stocking density of 8.8 kg-worm m⁻² fed on municipal solid waste (Pincince *et al.* 1981) and 0.25 g g-worm⁻¹ d⁻¹ for *E. fetida* (Hartenstein *et al.* 1981). However, due to variations in feedstock (and hence, its nutritional value), stocking density, earthworm sp., feeding regime and culturing conditions, such comparisons may be unreliable (*q.v.* Lofs-Holmin 1985).

Variations in processing were significantly different between 2-week feeding periods ($p < 0.0001$, two-way ANOVA), accounting for a larger proportion of the total variance (63.3%) than stocking density (11.7%). This was partly expected, as worm nutritional requirements were likely to have decreased during the experimental period. However, large reductions in processing ($p < 0.05$, Tukey-Kramer tests) at weeks 4 and 8 may suggest that the WPS had become less palatable to earthworms after 2 weeks storage.

Higher conversion ratios (see above) suggested higher stocking densities were more efficient at converting WPS into worm mass. Higher worm numbers processed WPS more rapidly, thus capturing proportionally higher levels of nutrition which may decrease with incubation time. Frederickson *et al.* (1997) found that the nutritional value of green-waste decreased proportionally to its organic matter content, which decreased rapidly with composting duration. Neuhauser *et al.* (1988) found that the nutritive value of sewage-sludge diminished rapidly when 'aged' at 20°C, again its nutritional value appeared to be closely related to organic matter content.

The effect of processing on the TOM content of WPS

The ingestion of WPS by *D. veneta* effected a significant ($p < 0.01$) mean loss in TOM of 45.8%ash compared to 30.1%ash for non-ingested WPS, a difference of 52%.

Increased stocking density had a slight effect on the TOM content of ingested WPS after 2 weeks (75.4%DM - 4 worms, 75.3% - 8 worms, and 75.0% 16 - worms) though this was not statistically significant ($p > 0.05$).

However, increased stocking density had a statistically significant effect on the TOM content of non-ingested WPS after 2 weeks ($p < 0.01$, two-way ANOVA); 80.1%DM – 4 worms, 79.6% - 8 worms, and 79.3% 16 – worms.

In both cases (ingested and non-ingested WPS), feeding-period accounted for a greater proportion of the total variance (88.4 to 95.5%). TOM contents of ingested and non-ingested WPS at weeks 4 and 8 were lower than for weeks 2 and 6. As fresh WPS was added at weeks 2

and 6, this suggested significant losses in TOM might have occurred during the cold storage (2-4°C) of WPS, demonstrating its readily biodegradable nature.

Hartenstein & Hartenstein (1981) reported a significant ($p < 0.05$) mean increase in TOM loss of 30% in activated sludge, affected by the presence of earthworms (*E. fetida*) over 4 weeks ('almost all sludge converted into castings'). From the same data (*op. cit.* 1981) it can be calculated that after 2 weeks the increase caused by the presence of earthworms was 57%, a similar figure to that observed for this study. In a study of the effect of earthworms of paper-sludge mixed with sewage, Elvira *et al.* (1996) results showed a 55% decrease in TOC in the presence of *E. andrei* and 43% decrease in its absence. This was equivalent to the earthworms causing a 28% increase in TOC loss (over 4 weeks), a similar finding to Hartenstein & Hartenstein (1981). Vincelas-Akpa & Loquet (1994) recorded TOC losses of 24.4% and 18.6% in the presence or absence of earthworms (*E. fetida/andrei*), respectively, after one month. This equates to an increase in TOC loss due to the worms presence of 31.2%; again this is very similar to other findings (*op. cit.* 1981 and 1996). However, in the study conducted by Elvira *et al.* (1996) only one replicate was used for each treatment; and no statistical comparisons were made between treatments by Vincelas-Akpa & Loquet (1994).

Aluminium content of earthworms

A change in opinion regarding the low potential for aluminium toxicity in ecosystems and animals has been suggested (Gromysz-Kalkowska & Szubartowska 1999). Rainbow trout exposed to chronic levels of aluminium are known to accumulate the element in heart and brain tissues, causing toxicity (Exley 1996). In a report concerning sources and the effect of aluminium on humans, it has been stated that ingestion is the largest source of aluminium for the general population (WHO 1997). Metal accumulation as a potential problem with using earthworms as a source of food, has been long identified (Sabine 1983).

Webb (1994) reported aluminium contents of between 0.2 and 7.4% for various fresh WPS; probably related to the variable dosage of coagulants used during waste water treatment (Smethurst 1992). Although this may be reduced by the use of polyelectrolyte coagulants (*op. cit.* 1992), these may also be toxic to earthworms. Lofs-Holmin (1985) found cationic polyelectrolytes [*e.g.*, poly-aluminium chloride, known to be contained within the WPS used in the current study] were highly toxic to *Allobophora caliginosa* in solutions of 10-50ppm. When culturing soil worms, Butt (1990) suggested that paper sludge from different origins might contain substances harmful to earthworms in general.

In the study described in this Chapter, final cultures of *D. veneta*, derived from large and small juveniles, showed signs of aluminium accumulation. Higher concentrations of aluminium in worms derived from a culture of small juveniles, appeared to be related largely to the higher

proportion of worm biomass gained using WPS as a feed, though statistical analyses were not possible.

The final aluminium concentration of worms derived from cultures of small juveniles (422 mg kg⁻¹) were similar to that observed for other earthworm species. Hartenstein & Hartenstein (1981) reported an increase in aluminium concentration from 160 mg kg⁻¹ to 430 mg kg⁻¹ in *E. fetida* cultured on activated sewage-sludge with a aluminium content of 8833 mg kg⁻¹.

Worms (*D. veneta*) purchased from two commercial suppliers, one based in the UK and one based in Hungary were shown to contain broadly similar levels of aluminium (576 and 478 mg kg⁻¹, respectively; data not shown). These suppliers were also known to use WPS as a feed during worm cultivation and storage.

Earthworms are known to accumulate high levels of heavy metals from various sources (Edwards & Bohlen 1992), for example, sewage sludge used as a feed (Hartenstein & Hartenstein 1981) or applied to the soil (Beyer *et al.* 1982). Although metal accumulation is not always lethal, sublethal effects of heavy metals are reported, such as reduced growth and cocoon production (Neuhauser *et al.* 1984). Although many studies have been conducted into the effects of heavy metals on various earthworm species (Edwards & Bohlen 1992), few have investigated aluminium. Aluminium is known to be more toxic than Cr³⁺ in some freshwater animals, such as planaria (Calevro *et al.* 1998).

A report on the toxic effects of aluminium on the earthworm *E. fetida* (Philips & Bolger 1998) showed that the effects of aluminium are dependent upon pH levels. They concluded that the lethal dose of aluminium was between 2000 and 4000 mg kg⁻¹ artificial soil at pH 4.2, but sublethal effects, such as inhibition of cocoon production, was observed between pH 4 and 7. At a pH of 6 and an initial aluminium concentration of 4970-5800 mg kg⁻¹ DM (1014-1508 mg kg⁻¹ WM), the WPS used in this study could have produced similar sublethal effects, as found during the experiments detailed in this Chapter. In addition, heavy metals increase in concentration during the degradation of WPS sludge (see Chapter 2), and their extractability may be increased by the action of earthworms (*q.v.* Elvira *et al.* 1995a). However, the effects of the form (salt) of aluminium, its combination with other substances, differences between earthworm species, degree of bioaccumulation, acclimatisation, and earthworm size/age are unknown. For example, Rundgren & Nilsson (1997) observed inhibition of the 'juvenile growth' of *Dendrodrilus rubidus* at relatively low aluminium concentrations (10-25 mg kg⁻¹ soil), using AlCl₃ as a source of aluminium. Kaplan *et al.* (1980) observed reduction in the growth in *E. fetida* cultured for 2 weeks on sewage sludge with an aluminium sulphate concentration of >1000 mg kg⁻¹ WM [Al concentration of only 158 mg kg⁻¹ WM], at a pH of 6.4.

With the increasing trend towards vermicomposting industrial sludges, there is a pressing need to fully investigate the bioaccumulation of PTEs into earthworm tissue. Published studies suggest that sublethal effects such as retarded growth and reproduction can occur at relatively low concentrations of certain metals. This study confirms that aluminium concentrations can be very high in some waste paper sludges and even at moderate, sublethal concentrations, worms cultured in WPS can accumulate aluminium.

3.5 Conclusions

The nutritional (physicochemical) composition of WPS appeared to be appropriate for earthworm culture, with suitable moisture, TOM, fibre, C:N ratio, salt (EC) and pH levels.

However, total worm biomass and individual biomass production were lower than comparative studies using WPS, but higher than studies using other wastes (*e.g.*, animal manure).

Although individual worm growth decreased with increased stocking density, total worm mass production was higher at higher densities, producing greater conversion ratios of WPS into worm mass. Worm cultures derived from large juveniles produced larger worms and showed lower levels of mortality.

High mortality, restricted growth and inhibited cocoon production was observed, mainly in stocking cultures of small juveniles. This was attributed partly to stocking density and, tentatively, to the degradation of the WPS feedstock during storage and incubation, and possibly toxicity caused by high levels of aluminium within the WPS.

Aluminium accumulation within *D. veneta* suggested worms cultured on WSP should not be utilised for animal feed, *e.g.* fish farming or angling bait. It is suggested that the effects of aluminium be investigated further in relation to the culture of earthworms on WPS sludges.

WPS processing and TOM losses were higher at higher stocking densities, and worm working appeared to affect TOM losses for ingested and non-ingested WPS. It is tentatively suggested that higher stocking densities may be more efficient at capturing the nutritional value of WPS, which may be higher at earlier stages of decomposition.

Overall, although the WPS used during the experiments appeared to be a highly suitable feedstock for vermicomposting, high worm mortality and limited growth and reproduction suggest that problems may be encountered in sustaining adequate worm populations in the long term. It is suggested that to ensure maximal WPS processing and TOM loss, high stocking densities of *D. veneta* should be employed. Furthermore, if larger *D. veneta* worms are used, earthworm mortality may be reduced and an increased individual worm size may be obtained. To maximise the reproduction of *D. veneta*, lower initial worm densities are suggested, and it is recommended that this species is cultured on alternative substrates to WPS.

4. Large scale composting of a commingled primary and secondary waste paper sludge using an open-air, mechanically turned windrow system.

4.1 Introduction

In Chapter 2 it was seen that the windrow-composting of particular primary and secondary WPS types required amendment with mineral nutrients or bulking agents, respectively, to achieve adequate levels of microbial activity. Equally, mixing the two sludges (2:1 by volume) also allowed composting to take place and achieved satisfactory results. When the sludges were mixed and additional liquid fertiliser applied, there appeared to be inhibition of composting compared to omitting the fertiliser. As a result of this research it may be concluded that further investigation into composting commingled primary and secondary sludges would be justified, and that a principle focus of investigation should be increasing aeration of the waste mix, rather than nutrient addition. Building on this large-scale research, a survey of selected paper mills suggested that primary and secondary sludges were often commingled prior to disposal. Therefore, although readily available, this proves problematic to landfill. This Chapter investigates the effectiveness of mechanically turned windrow composting of commingled primary and secondary waste paper sludge without the use of nutritional amendment. The effect of incorporating a predominantly structural amendment (wood chip bulking-agent), and of increasing the rate of agitation (turning frequency), on the windrow-composting of the WPS were investigated.

A large scale windrow system was set up to investigate the following research questions:

1. Does the addition of a bulking agent enable the efficient composting of WPS under a relatively infrequent turning regime?
2. Does a relatively frequent turning regime enable the efficient composting of WPS without the aid of a bulking agent?

The following hypotheses were used as the rationale behind the investigation:

1. The addition of a bulking agent to WPS will increase the porosity of the windrow, facilitating the passive diffusion of respiratory gases into and out of the WPS; allowing increased microbial activity and accelerated WPS decomposition through increased aeration (O_2).

2. Increased agitation of WPS will relieve compaction, increasing windrow porosity, and actively introducing and removing respiratory gases from the WPS; allowing increased microbial activity and accelerated WPS decomposition through increased aeration (O_2).

A particular aim of this work was to identify the method of windrow composting which proved most successful in terms of WPS decomposition and stabilisation in order to make a objective comparison with optimal vermicomposting (Chapter 5). The final windrow composted WPS was also used in plant growth trials to investigate differences in windrow composted and vermicomposted WPS products (Chapter 6).

4.2 Materials and methods

4.2.1 Feedstock preparation and windrow formation

A feedstock of commingled primary and secondary WPS was obtained from a packaging manufacturer (SCA). The WPS consisted of a mixture of fibrous primary clarifier residues and secondary (activated) sludge, belt pressed into a sludge cake with a solids content of 20-25%. The WPS was divided into three equal fractions, each containing approximately 5 tonnes of material.

One fraction of WPS was mixed with birch wood chips in a ratio of 1:1 by volume (bulking-agent [BA] : windrow). The the other two fractions were unamended.

Each fraction was formed into a windrow (height - 1.5 m; width - 3 m; length - 4 m for unamended WPS, and 6 m for amended WPS).

4.2.2 Process Control

Windrow turning regime

All windrows were mixed and turned using a tractor-mounted loading shovel. Each turning consisted of the movement of a windrow to an adjacent position, followed by its relocation. This process ensured thorough agitation and mixing of the feedstock.

WPS was composted for 12 weeks, consisting of an 8 week active phase (with turning) followed by a four week maturation phase (without turning).

The windrow containing WPS amended with wood-chips (BA) was turned once per week. One windrow containing non-amended WPS was turned three times per week (frequently-turned [FT] windrow); and the second windrow containing non-amended WPS, was turned once per week (control windrow).

Moisture content

No additional water was required initially or during the experimental period. Windrows were lightly covered with ventilated plastic sheeting during periods of excessive rainfall to prevent saturation of the windrows. No significant leaching was observed.

4.2.3 Process monitoring

Windrow temperature

Windrow temperatures were recorded using 8×1 m temperature probes connected to a digital data-logger (IceSpy[®], © Silvertree Engineering Limited). Four probes were placed at a height of 1 m and four at 0.5 m from the base of the windrow; two probes were inserted to a depth of 0.3 m and two to a depth of 0.6 m, in each case. The positioning of probes was kept consistent between recording periods, and between windrows. Data-loggers recorded the temperature of each probe at 10 minute intervals, and average weekly temperatures were calculated from average daily readings.

Windrow Gas analysis

Each experimental windrow's internal gas composition was analysed using a 1 m steel-tubing (1.5cm bore) probe connected to a Dräger Multiwarn II gas analyser with built-in digital data-logger. Gas was analysed for oxygen (O_2), carbon dioxide (CO_2) and ammonia (NH_3) contents; O_2 and NH_3 were measured by electrochemical sensors and CO_2 was measured by an infra-red light sensor. All recorded data were downloaded to a computer using GasVision evaluation software (© 1996 Drägerwerk AG).

During gas analysis, the probe was placed at 3 different positions along each windrow, at a height of 1 m from the base of the windrow. At each position the probe was inserted to a depth of 0.3 m and 0.6 m. For each of the 6 gas recordings per windrow, the probe was left inserted for 2 minutes to ensure a stable gas reading was achieved.

The positioning of probes was kept as consistent as possible between recording periods, and windrows.

Sampling regime

Samples of WPS were collected at weeks 0 (fresh) 1, 2, 3, 4, 6, 8 and 12. Four samples were taken from each windrow at each sampling period. Windrows were marked out into four equal sections, and from each section 10 x 1 litre portions of the composting material were taken at random from a depth of approximately 0.3 m from the windrow's surface. Samples were aggregated and mixed thoroughly. From each 10 litre aggregate sample, a 2 litre sub-sample of WPS was taken. All traces of bulking-agent were removed by hand sorting. The 4 samples

from each sampling period were kept under frozen storage in hermetically sealed plastic bags until analysis. Samples were taken immediately after turning to increase the homogeneity of the sampled material.

Physicochemical analyses

Moisture content

All samples were analysed for moisture content (% wet mass). Moisture content was estimated gravimetrically, as the loss in mass of a 100g portion of sample dried in a fan assisted drying oven at 105 °C for 24 hours.

Total organic matter and carbon contents

Total organic matter content (TOM) was determined for all samples. A 100g portion of each sample was dried at 105°C for 24 hours, and milled to a <1 mm particle size using a Messerm-125H knife-mill. Portions (approximately 5g) of the prepared samples were placed in ceramic crucibles, and ignited in a muffle furnace. TOM was calculated as the loss in mass (volatile solids) on ignition at 550°C for 2 hours (FCQAO, 1994).

Total organic carbon (TOC) content was calculated as 55% of the TOM (Adams *et al.*, 1951); this falls within a range of between 45 and 60% recorded for some recycled WPSs (Bellamy *et al.* 1995).

*Water extractable ions **

Water soluble ammonium (NH_4^+), nitrate (NO_3^-), phosphate (PO_4), potassium (K^+) and magnesium (Mg^{2+}); pH and electrical conductivity (EC) were determined for fresh feedstock and samples taken after 2, 4, 6 and 8 weeks active composting; and after maturation (week 12).

*Total macro-nutrient contents **

Total (Kjeldahl) nitrogen (N), total phosphorus (P) and total potassium (K) contents were determined for fresh feedstock and samples taken after 2, 4, 6 and 8 weeks active composting; and after maturation (week 12).

* Analyses were conducted by 'Levington Agriculture', Levington Park, Ipswich, Suffolk, UK; using standard ADAS procedures.

Mathematical and Statistical analysis

Relative nutrient losses (correcting for dry mass losses)

Nutrient losses were corrected for the concentration effect due to the reduction in organic matter during composting. Initial ash content (%DM) was used as a baseline, assumed to remain at a constant total mass (Stentiford & Pereira Neta 1985).

Using the theory of ash conservation, dry mass (DM) contents can be calculated using the following equation:

$$DM_t = TOM_t + ash_0 \quad (1)$$

TOM = total organic matter content after composting duration (t); Ash_0 = initial ash content (adapted from Genevini *et al.* 1996).

Thus nutrient losses may be corrected for loss of DM using the conservation of ash:

$$X\%_{ASH} = \left\{ 1 - \left(\frac{\%X_t \times \%ash_0}{\%X_0 \times \%ash_t} \right) \right\} \times 100 \quad (2)$$

X = nutrient; $\%ash_0$ = initial ash content (%DM); $\%X_0$ = initial nutrient content (%DM); $\%ash_t$ = final ash content (%DM) after composting duration (t); $\%X_t$ = final nutrient content (%DM) after composting duration (t) (adapted from Bernal *et al.* 1996).

Non-linear regression of TOM with composting duration

Curves were fitted to TOM (%DM) data against composting duration, using a one-phase exponential decay model:

$$TOM (\%DM) = a \times e^{(-\lambda x)} + c \quad (3)$$

a = total loss in TOM (%DM) (initial TOM – c); c = final value of TOM (%DM);

λ = decay constant; Half-life (T) = $(\ln 2)/\lambda$.

Curves were fitted using SPSS[®] statistical analysis software package (© SPSS Inc.), using the Levenberg-Marquardt method of estimation (sum-of-squares minimisation).

Though usually associated with radioactive decay, the one-phase decay model was based on scientific assumptions felt appropriate for modelling organic matter decomposition, *i.e.*, a decelerating rate of decay before reaching an essentially constant final value. The model was used to estimate values of total loss of TOM (%DM); final value of TOM (%DM); and the rate at which these losses occurred (λ & T).

Statistical differences between decay curves were calculated from the means and asymptotic standard errors (se) of variables (a , λ , and c). A one-way analysis of variance (ANOVA) was used to find statistical differences between the variables of different experimental windrows; statistical differences between specific pairs of windrows were analysed using post ANOVA, Tukey-Kramer multiple comparison tests.

Statistical analysis of acidity/alkalinity

Statistical differences in acidity/alkalinity were calculated using $[H^+]$; pH was converted into $[H^+]$ using equation 4.

$$[H^+] = 10^{-pH} \quad (4)$$

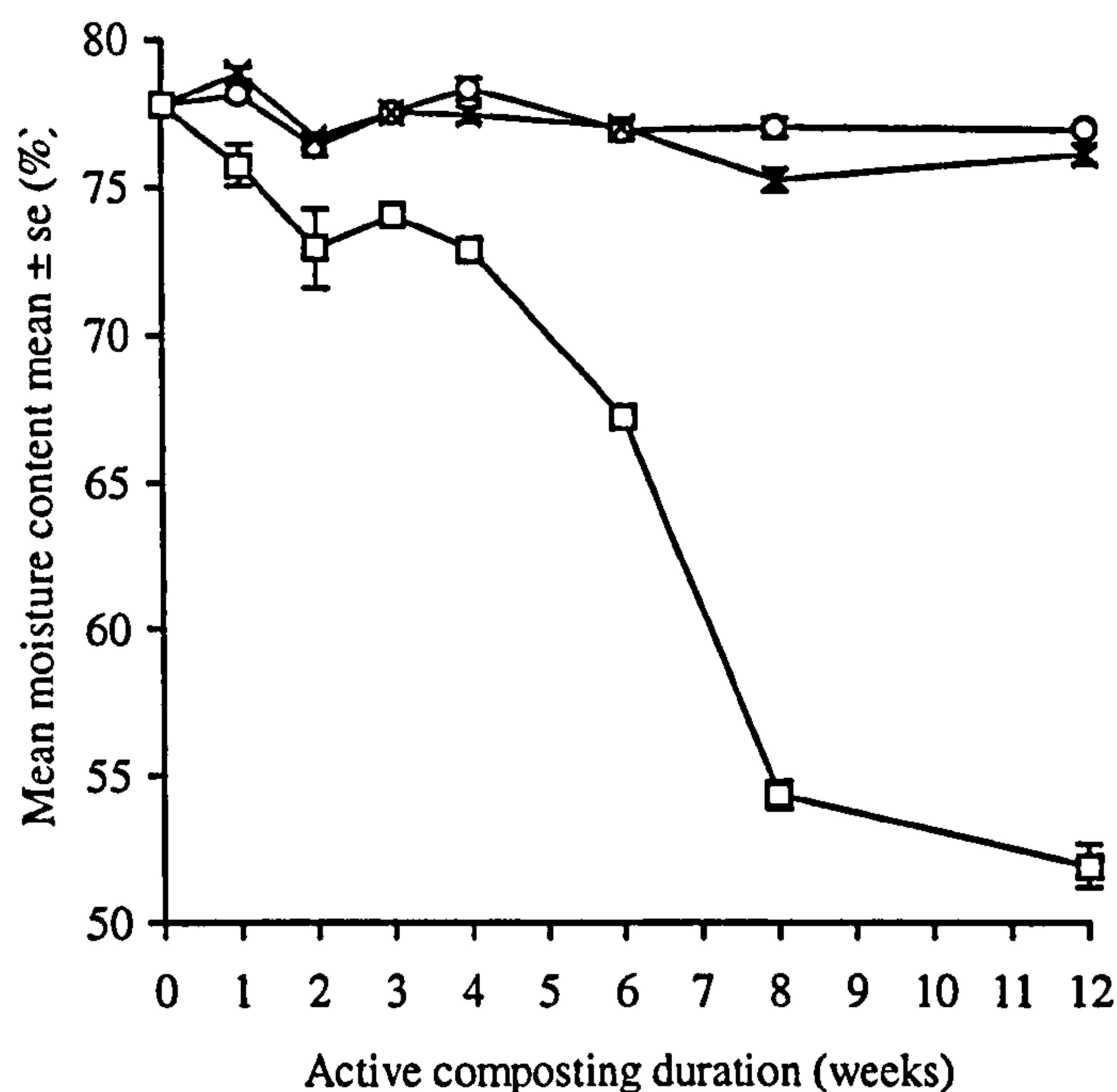
General statistical analysis

Statistical analyses of differences between means were performed using standard two-tailed Student t-tests, assuming equal standard deviations of means. Modified t-tests or other statistical methods utilised are stated next to p-values.

4.3 Results

4.3.1 Moisture

Figure 23 – Changes in moisture content; Windrow treated with bulking agent (BA) [—□—]; Windrow treated with frequent turning (FT) [—×—]; Control windrow [—○—].



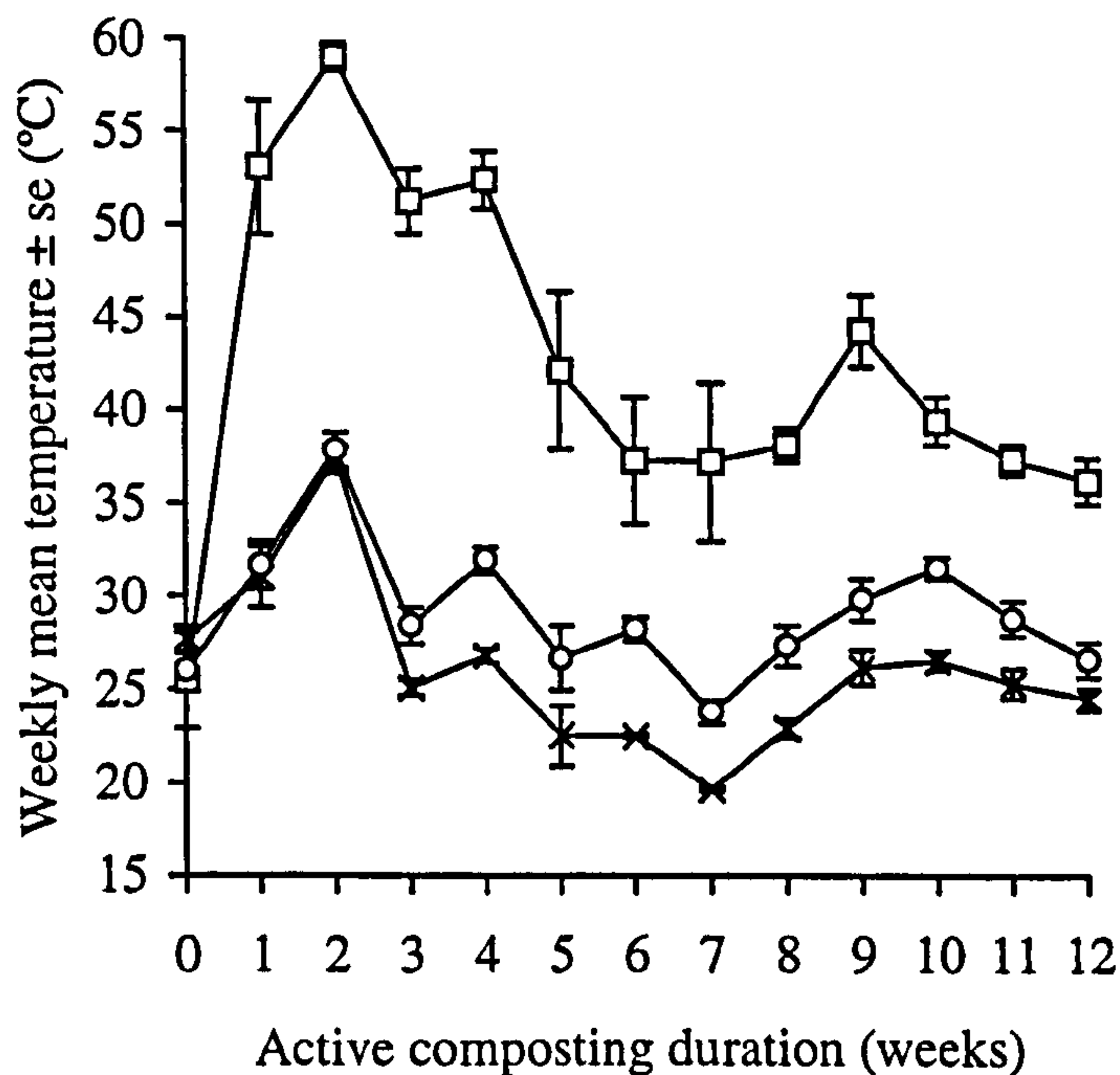
The initial moisture content of WPS within the windrows was 77.8%. No additional water was applied at the start of composting.

During the 12 week experimental period mean moisture content varied little within the frequently turned (FT) and control windrows; fluctuating between 76.4 to 78.4% for the FT windrow and between 75.3 to 78.9% for the control.

The mean moisture content of the windrow treated with a bulking agent (BA) was significantly below the FT and control windrows ($p < 0.01$) from week 1 onwards. The moisture content of the BA windrow fell significantly below its initial moisture content ($p < 0.05$) from week 2 onwards, falling to 51.9% by week 12.

4.3.2 Temperature

Figure 24 - Changes in windrow temperature; Windrow treated with bulking agent (BA) [-□-]; Windrow treated by frequent turning (FT) [-×-]; Control windrow [-○-].

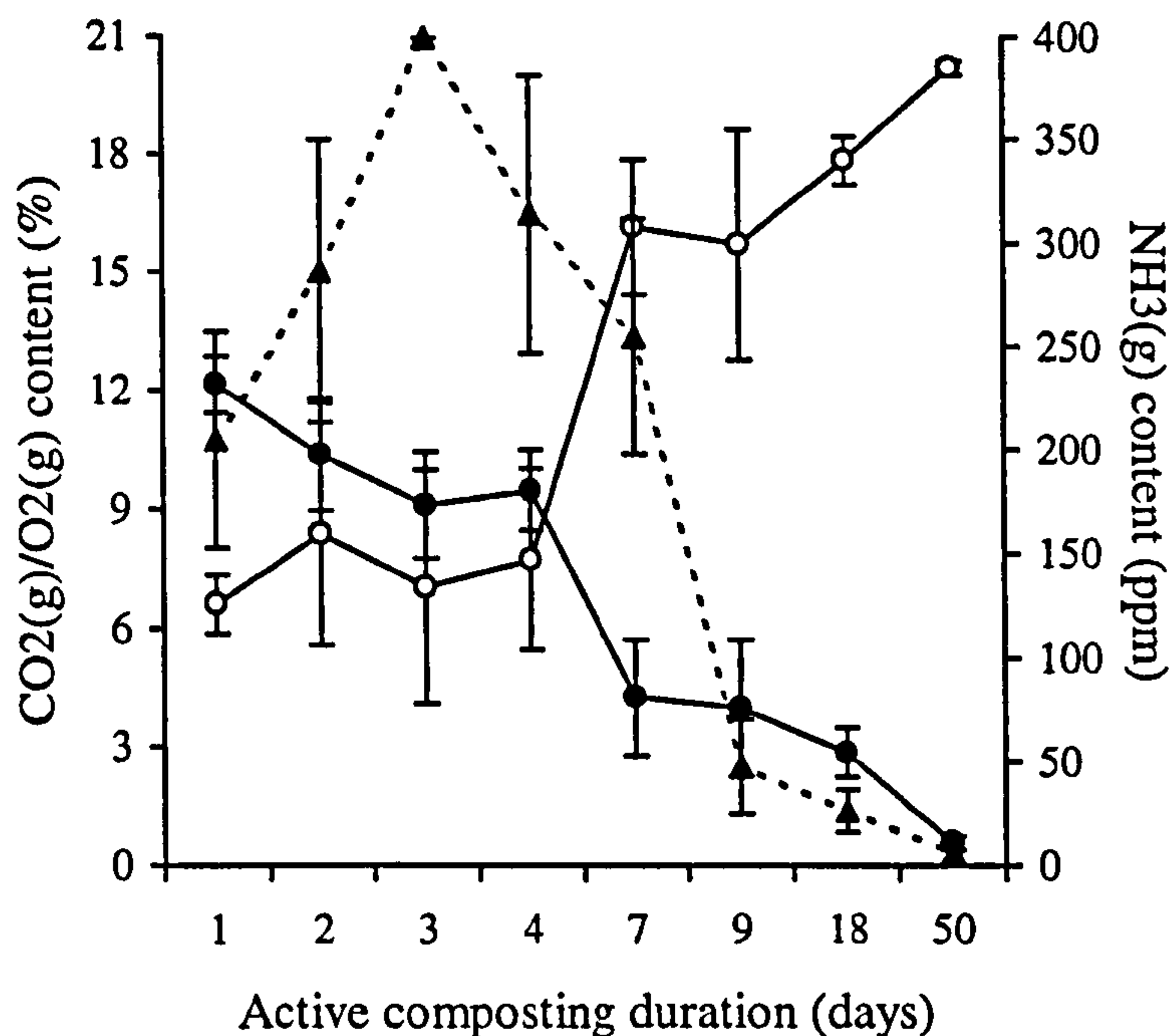


Temperatures within the FT and control windrows showed a similar increase over the first 2 weeks of composting, reaching maximum temperatures of 37.9°C and 37.3°C respectively. After week 2, temperatures within the FT and control windrows fell, and fluctuated between 23.8 to 31.9°C and 19.6 to 26.8°C, respectively, thereafter. The FT windrow temperature remained between 3.3 to 5.8°C higher than the control ($p < 0.05$), until week 12 where there was no statistically significant difference.

The BA windrow temperature was recorded at between 51.3 to 59.1°C for weeks 1 to 4 of the active composting period. Thereafter the BA windrows temperature began to fall rapidly, reaching a minimum temperature of 36.2°C by week 12. The BA windrow temperature was significantly higher than the FT windrow or control at all points during the 12 week experimental period ($p < 0.01$); an average of 22°C higher than the FT windrow and 24°C higher than the control windrow during weeks 1 to 4.

4.3.3 Gas analyses

Figure 25 - Composition of interstitial gases within the BA windrow; Carbon dioxide [—●—]; Oxygen [—○—]; Ammonia [—▲—].



The composition of gas within the interstices of the BA windrow for the first 4 days of composting consisted of 9.1-12.2% carbon dioxide (CO₂) and 6.6-8.4% oxygen (O₂). At days 7 and 9 CO₂ levels had fallen significantly ($p < 0.001$) to 4.0-4.2%, and O₂ levels had risen significantly ($p < 0.001$) to 15.7-16.2%. At day 18 of active composting CO₂ and O₂ levels had not changed significantly ($p > 0.05$) from day 9. After 50 days of active composting (7 weeks) CO₂ levels had fallen significantly ($p < 0.001$) to $0.6 \pm 0.2\%$ and O₂ had risen significantly ($p < 0.05$) to $20.2 \pm 0.2\%$; at this stage neither CO₂ nor O₂ were significantly different from ambient levels (0.03% and 20.9% respectively).

CO₂ levels within the BA windrow for days 1-4 were significantly below levels observed within the FT and control windrows at these designated sampling points ($p < 0.001$, one-way ANOVA) with mean differences of 6.1% and 6.9% respectively; O₂ levels were significantly higher ($p < 0.01$, ANOVA) with mean differences of 8.0% and 5.8% respectively.

CO₂ levels within the BA windrow for days 7, 9, 18 and 50 were significantly below levels observed within the FT and control windrows at these sampling points ($p < 0.001$, ANOVA) with mean differences of 12.7% and 14.6% respectively; O₂ levels were significantly higher ($p < 0.001$, ANOVA) with mean differences of 12.4% and 13.2% respectively.

For the bulking agent windrows ammonia (NH₃) levels appeared to peak during the first week of composting, reaching levels in excess of the gas sensor recording range (400ppm) at day 3.

By day 9 NH₃ level fell significantly ($p < 0.001$) to 47.5 ± 22.8 ppm. At day 50 the NH₃ level was only 6.2 ± 0.4 ppm, which was not significantly different from ambient levels (0 ppm).

Figure 26 – Composition of interstitial gases within the FT windrow; Carbon dioxide [—●—]; Oxygen [—○—].

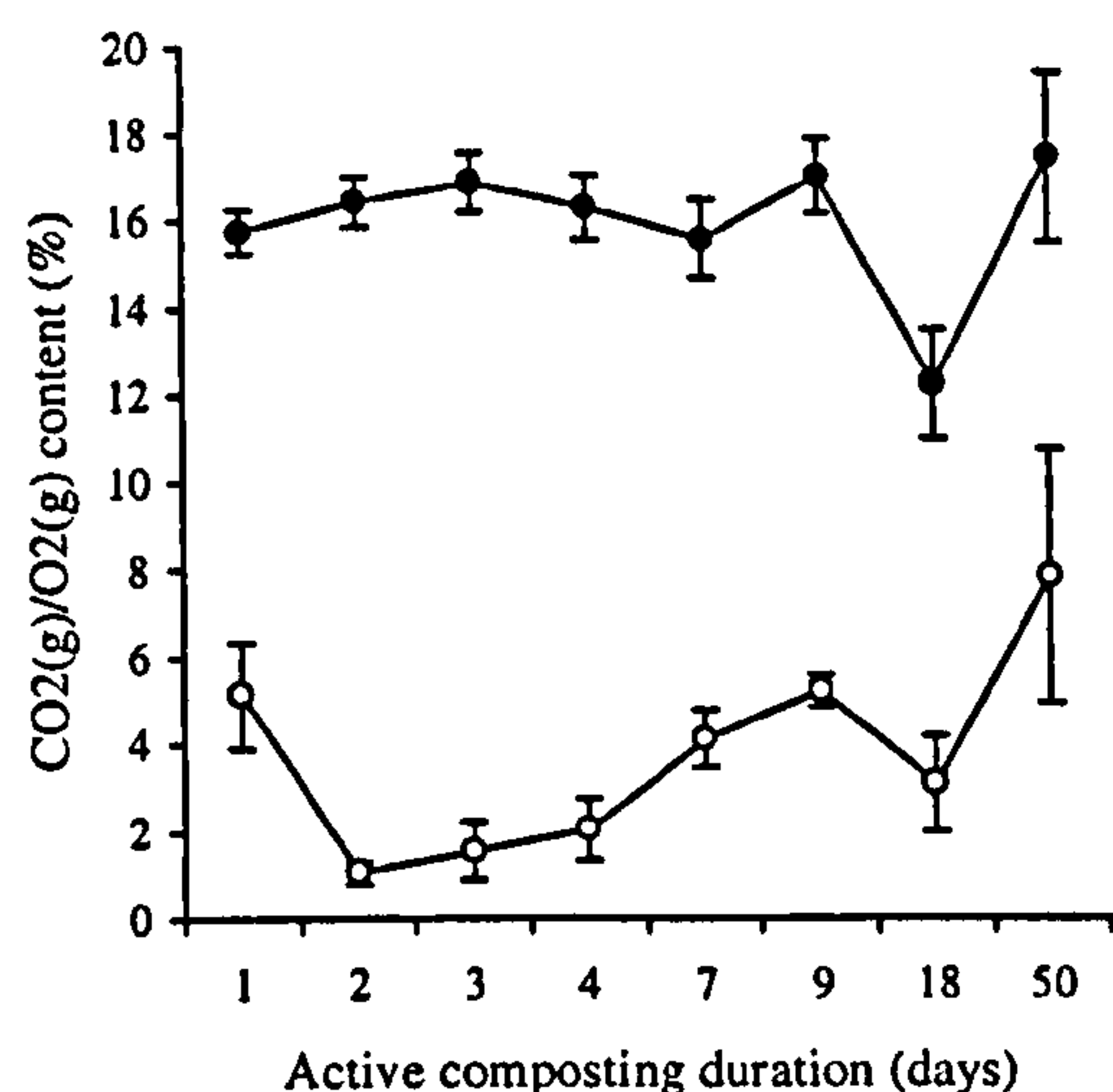
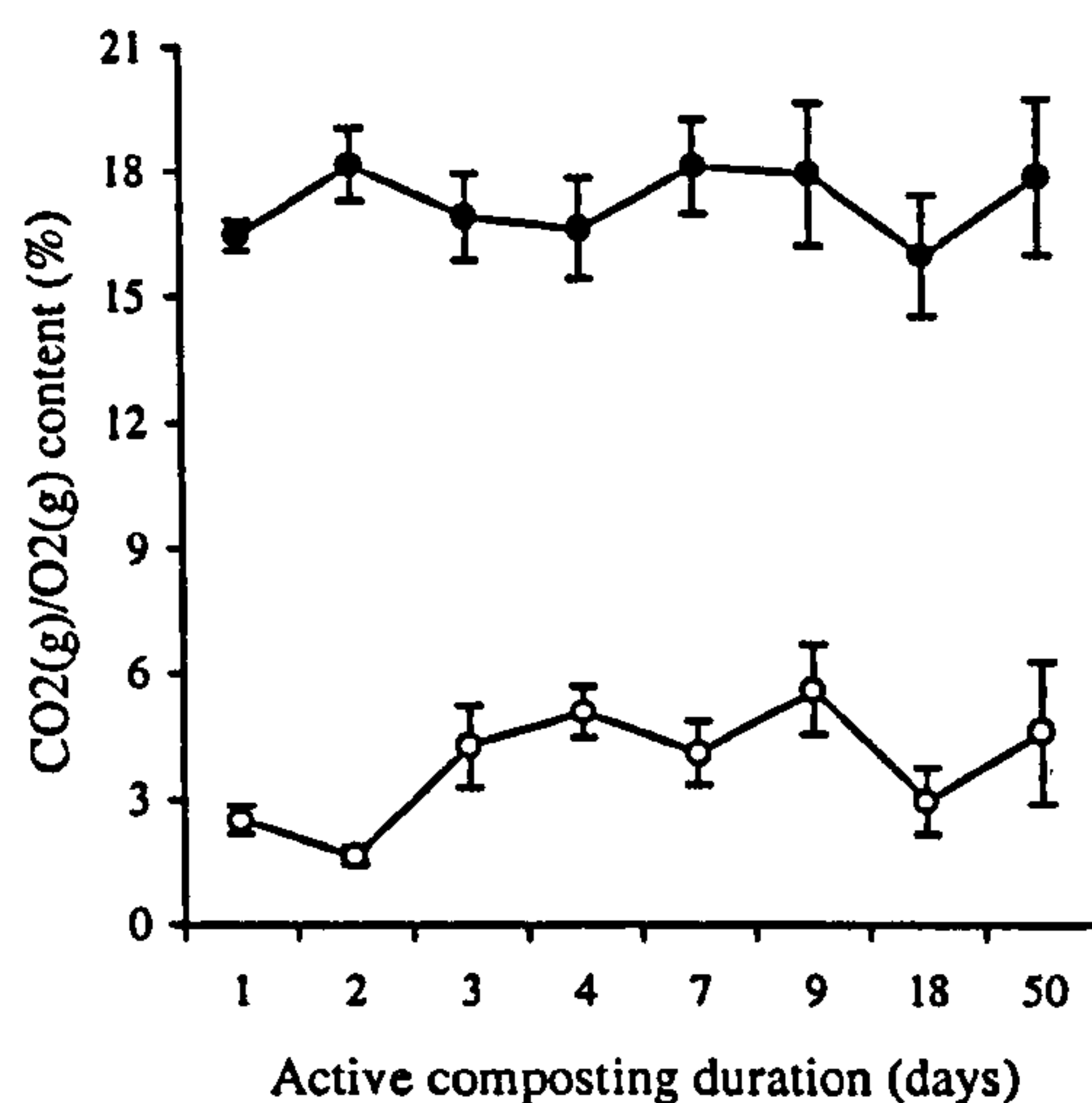


Figure 27 – Composition of interstitial gases within the control windrow; Carbon dioxide [—●—]; Oxygen [—○—].



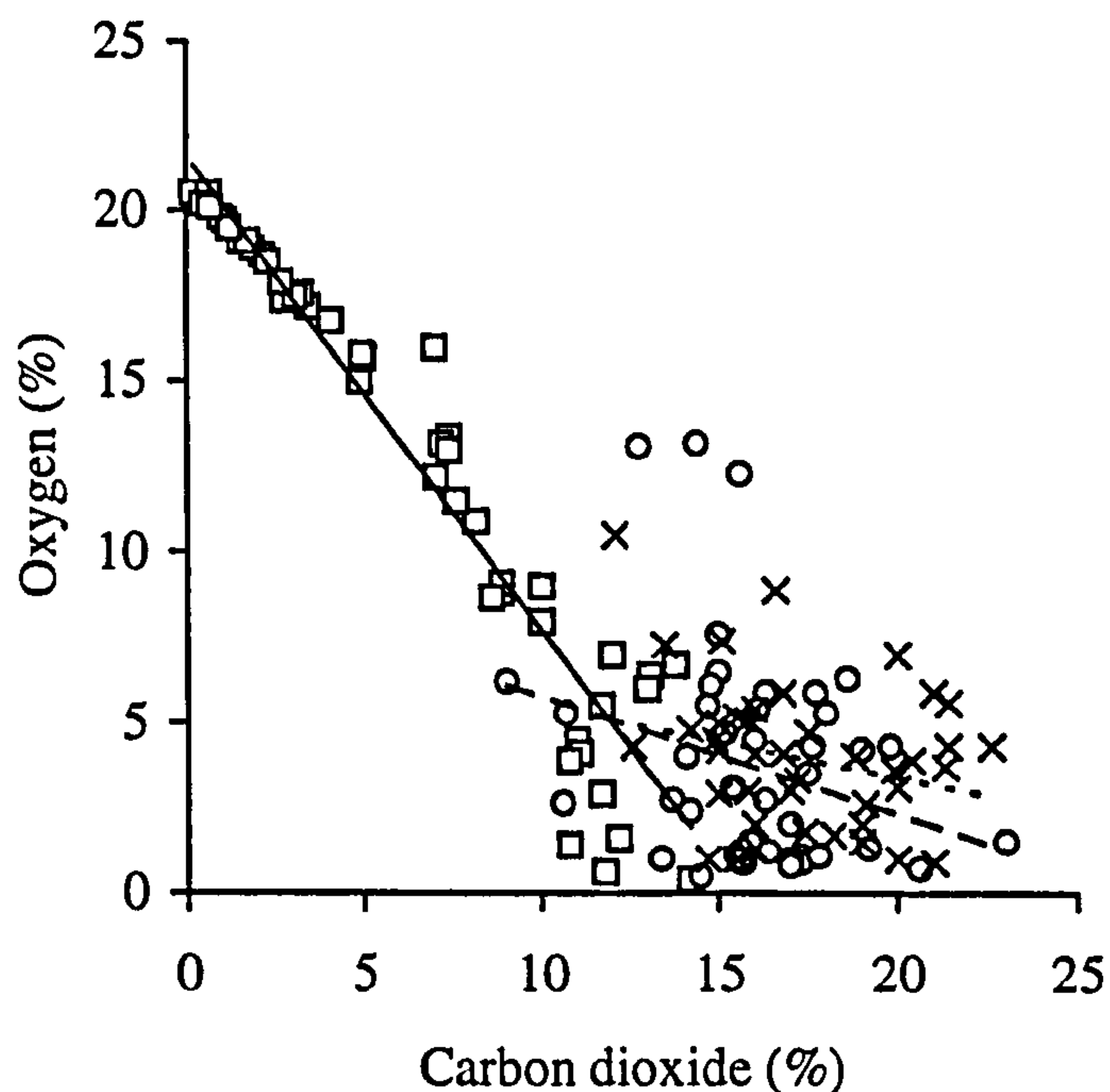
The composition of gas within the FT windrow and control windrow showed very high CO₂ levels at all sampling points, falling between 12.3 to 17.5% and 16.0 to 18.1% respectively; O₂ levels were correspondingly low falling between 1.1 to 7.9% and 1.6 to 5.6% respectively. During gas sampling and analysis, the probe would often take up water, indicating water-logged conditions within both windrows.

Mean CO₂ and O₂ levels within the FT windrow did not differ significantly from the control windrow over all designated sampling points ($p > 0.05$, one-way ANOVA).

NH₃ levels (not plotted) exceeded the recording range of the gas sensor (400 ppm) at all sampling points for both windrows.

Figure 28 - Relationship between carbon dioxide and oxygen content;

Windrow treated with bulking agent (BA) [\square , linear fit = —]; Windrow treated by frequent turning (FT) [\times , linear fit — — —]; Control windrow [\circ , linear fit = - - - -]



The composition of gas within the BA windrow showed a very significant negative correlation between O_2 and CO_2 levels (Pearson $r = -0.96$; $p < 0.0001$; $R^2 = 91.9\%$). However, at high carbon dioxide levels, as in the case of FT and control windrows, no significant relationship between O_2 and CO_2 was observed (FT windrow: Pearson $r = -0.27$; $p > 0.05$; $R^2 = 7.5\%$; control windrow: Pearson $r = -0.26$; $p = 0.0952$; $R^2 = 6.6\%$).

Analysis of all data points from all three experimental windrows still revealed a significant negative correlation between O_2 and CO_2 (Pearson $r = -0.86$; $p < 0.0001$; $R^2 = 74.6\%$); linear regression gave a line with a y-intercept = 18.5%, x-intercept = 20.9%, and slope = -0.89 (significantly different from zero $p < 0.0001$). However, when all data points $\geq 7.0\%$ CO_2 were excluded from the analysis a much stronger correlation was revealed (Pearson $r = -0.99$; $p < 0.0001$; $R^2 = 98.0\%$); with a linear regression line: y-intercept (O_2) = 20.8%, x-intercept (CO_2) = 19.7%, and slope = -1.06 ($p < 0.0001$). A slope of -1 suggests that oxygen consumption and carbon dioxide production were directly inversely proportional. This is in agreement with Schulze (1960) who states a respiratory quotient of one, that is: CO_2 produced/ O_2 consumed = 1.

4.3.4 Total organic matter

Table 22 – Cumulative loss of TOM (%ash*); mean±se.

Week	BA	FT	Control
1	32.6±1.2	25.9±1.1	24.4±0.3
2	42.9±1.9 ^a	30.9±1.3 ^b	32.2±1.7 ^b
3	50.4±0.1 ^a	35.7±0.9 ^b	35.8±1.0 ^b
4	56.4±0.5 ^a	43.8±0.6 ^b	38.3±0.9 ^c
6	61.8±1.2 ^a	47.7±0.3 ^b	43.4±0.9 ^c
8	70.4±0.9 ^a	55.3±0.6 ^b	51.8±0.4 ^c
12	74.1±0.7 ^a	59.7±0.5 ^b	56.6±0.4 ^c

* for calculations see section 2.2.4.

^{abc} different letters indicate significant differences (p<0.05, post AVOVA, Tukey-Kramer multiple comparisons test).

Table 23 – Non-linear regression of TOM content (%DM) vs. composting duration; mean±se.

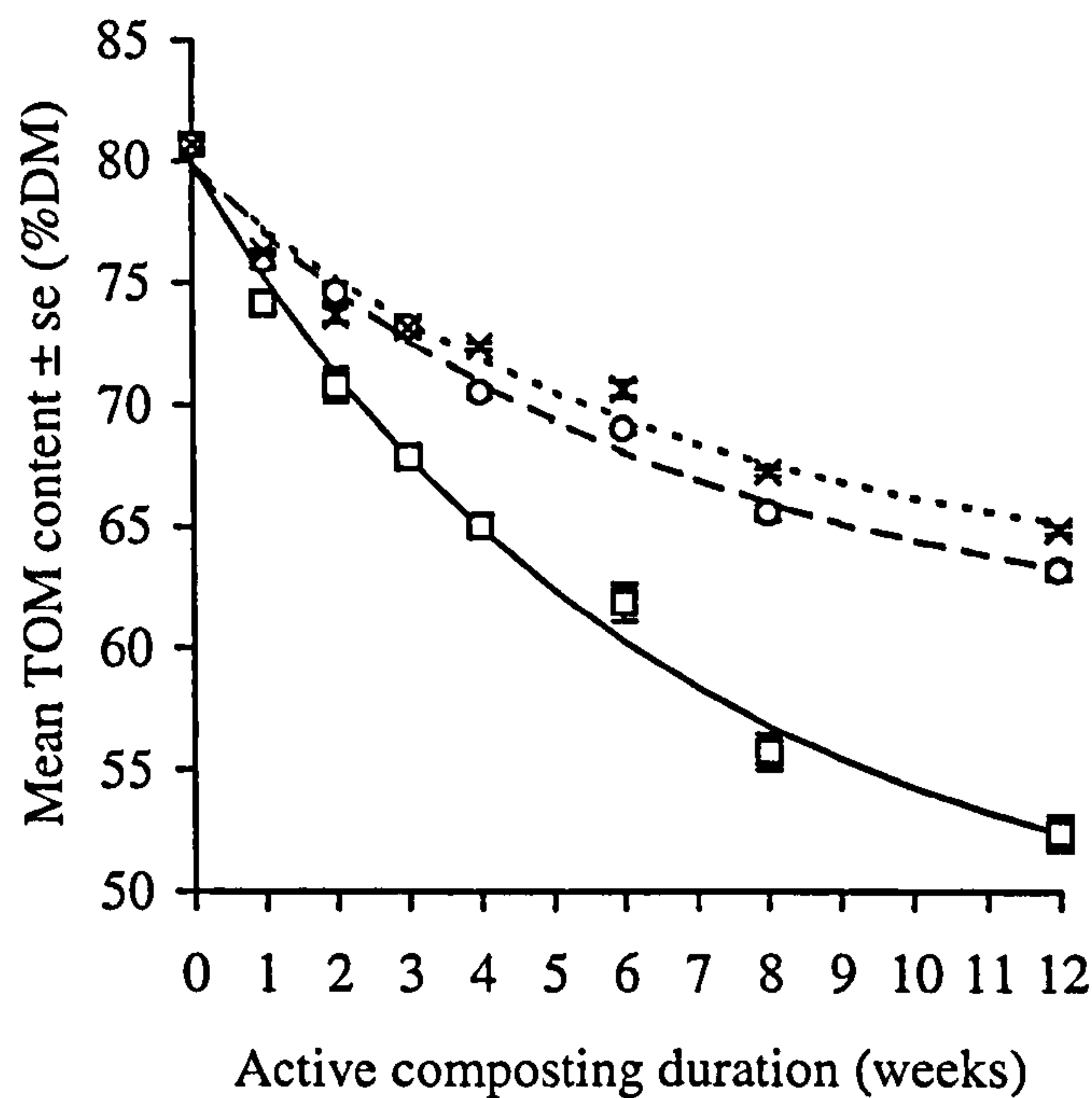
Variable	BA	FT	Control
Total TOM loss (a)	32.79±1.63 ^a	19.96±1.06 ^b	17.35±1.27 ^c
Final TOM (c)	47.07±1.80 ^a	59.97±1.17 ^b	62.12±1.14 ^b
Decay rate (λ)	0.152±0.017 ^a	0.150±0.018 ^a	0.145±0.023 ^a
Half-life (T) weeks	4.57±0.51 ^a	4.63±0.55 ^a	4.77±0.76 ^a
Rate of 50% of total TOM loss (TOM %DM wk ⁻¹)	3.59±0.29 ^a	2.15±0.19 ^b	1.84±0.21 ^b
Measures of goodness of fit			
R ²	98.0%	97.7%	96.2%
S _{x,y} (TOM %DM)	1.33	0.85	0.96

^{abc} different letters indicate significant differences (p<0.05, Tukey-Kramer multiple comparisons test).

R² = fraction of total variance in experimental data explained by the model.

S_{x,y} = standard deviation of points from the model line-of-best-fit. (Motulsky 1996)

Figure 29– Changes in TOM content (%DM) with composting duration; Windrow treated with bulking agent (BA) [□, best fit = —]; Windrow treated by frequent turning (FT) [×, line of best fit - - -]; Control windrow [○, line of best fit - - - -].



Results of non-linear regression show that the one-phase-exponential decay model provides a good fit for the TOM %DM data (Table 23), with >96% of data variance explained by the model, and a standard deviation of data from the model of <1.4%DM TOM.

The model suggests that the total loss in TOM %DM predicted by the model was not significantly different ($p>0.05$) between the FT and control windrow (see Table 23), whereas final TOM %DM levels were statistically different ($p<0.05$). However, the mean difference in final TOM %DM predicted by the model was only 2.15 %DM.

The BA windrow showed a significantly greater total TOM %DM loss and lower final TOM %DM predicted by the model than both the FT and control windrows. Mean differences in final TOM %DM predicted by the model between the BA and FT and control windrows were 12.9 and 15.1 %DM respectively.

The rate at which total TOM losses occurred, *i.e.* the rate at which the decay curves reach a plateau, did not differ significantly between windrows. However, due to the much greater total TOM loss that occurred in the BA windrow, actual losses of TOM %DM per unit time would have been higher. This is demonstrated by proportional decay values (Table 23) which were calculated as half the total TOM loss predicted by the model divided by half-life. As expected

this shows a significantly higher rate of decay (TOM %DM wk⁻¹) within the BA windrow ($p<0.01$), with no significant difference between the FT windrow and the control.

Cumulative losses of TOM corrected for ash (%ash) (Table 3) support the findings of non-linear analysis, showing significantly greater losses within the BA windrow from week 3 onwards, and no significant differences between the FT and control windrows until after week 3. In 8 weeks of active composting the BA windrow lost 70.4 %ash (25.2%DM), very significantly higher ($p<0.001$) than the FT windrow which lost 55.3%ash (15.4.%DM), and the control windrow which lost 51.8%ash (13.7%DM).

Further mean reductions, during the 4 week maturation period (weeks 8 to 12), were 3.7%ash (3.4%DM) in the BA windrow, 4.4%ash (2.4%DM) in the FT windrow, and 4.8%ash (2.4%DM) in the control. As predicted by the model, the total losses of TOM (after 12 weeks) were significantly higher within the BA windrow ($p<0.001$) than the FT windrow, and significantly higher in the FT windrow ($p<0.01$) than the control windrow.

4.3.5 Nitrogen content

Table 24 – Change in the various forms of nitrogen content*

Windrow	Sample (week)	TKN %	N loss %ash	TOC %	C/N ratio :1	Water soluble		NH ₄ ⁺ /N O ₃ ⁻ ratio :1
						NH ₄ ⁺ mg/kg	NO ₃ ⁻ mg/kg	
Fresh	0	2.70±0.06	n/a	44.6±0.1	16.5±0.5	686±78	19±1	36±6
BA	2	1.75±0.05	57.7±1.4	39.0±0.4	22.2±0.8	3269±441	23±2	144±38
	4	1.81±0.06	63.7±1.7	35.8±0.1	19.8±0.7	2780±143	20±1	141±16
	6	1.79±0.06	67.0±1.8	34.0±0.4	19.0±0.9	151±41	16±1	10±4
	8	1.89±0.05	70.0±1.8	30.7±0.4	16.3±0.7	213±55	113±51	2±2
	12	2.13±0.07	68.5±1.9	28.8±0.4	13.5±0.6	6±6	308±48	0±0
FT	8	2.17±0.05	55.5±1.3	36.1±0.2	16.6±0.5	6136±580	15±1	411±67
Control	8	2.49±0.04	46.4±0.9	37.0±0.1	14.8±0.3	5837±310	14±0	428±32

*All figures expressed in terms of dry mass where applicable.

During 8 weeks active composting the windrow with bulking agent (BA) amendment lost 70.0±1.9%ash of its initial level of TKN (p<0.01). Most (57.7%ash) of this nitrogen loss occurred during the first 2 weeks of composting, amounting to 84% of the total nitrogen loss. During 4 weeks maturation (weeks 8 to 12) there was no significant change in TKN (%DM), and no significant further loss in TKN (%ash).

A large net formation of ammonium (NH₄⁺) was observed within the BA windrow during the first 2 weeks of composting, amounting to 2583 mg kg⁻¹ (211±26%ash). By week 6 the level of NH₄⁺ had fallen significantly from weeks 2 and 4 (p<0.001), a decrease of 89±17%ash compared to fresh WPS. By week 12 practically all NH₄⁺ had decreased by 99.6±5.7%ash.

The level of nitrate (NO₃⁻) appeared to rise within the BA windrow by 94 mg kg⁻¹ by week 8, however due to large variation within the data this was not statistically significant (p>0.05). After 4 weeks maturation (by week 12) NO₃⁻ had risen significantly from week 8 (p<0.05), a mean of 289 mg kg⁻¹ above that present in fresh WPS (p<0.01).

At week 12, a low level of NH₄⁺ in the BA windrow coincided with an increased level in NO₃⁻, giving a low NH₄⁺/NO₃⁻ ratio of 0.021:1±0.004.

A significant reduction in C/N ratio from 16.5:1±0.4 to 13.5:1±0.6 within the BA windrow (p<0.01) over the 12 week experimental period, indicated a relatively higher loss of carbon than nitrogen (mean C loss = 74.1%ash, mean N loss = 68.5%ash).

During 8 weeks active composting the frequently turned (FT) and control windrow lost $55.5 \pm 1.3\%$ and $46.4 \pm 0.9\%$ of the initial level of TKN respectively ($p < 0.001$). Both these figures are significantly below that observed for the BA windrow ($p < 0.001$). The loss of nitrogen from the FT windrow was significantly greater than the control ($p < 0.01$).

A large net formation of ammonium (NH_4^+) after 8 weeks was observed within both the FT and control windrows, amounting to 5450 mg kg^{-1} ($394 \pm 41\%$) and 5151 mg kg^{-1} ($394 \pm 33\%$) respectively, with no significant difference between them ($p > 0.05$). These figures were significantly ($p < 0.001$) above the initial level (fresh WPS) of NH_4^+ , and significantly above the value observed for the BA windrow at week 8 ($p < 0.001$).

The level of nitrate (NO_3^-) within the FT and control windrows by week 8 showed no statistically significant difference from their initial levels ($p > 0.05$).

Very high levels of NH_4^+ within FT and control windrows (week 8), coinciding with no increase in NO_3^- level, resulted in very high $\text{NH}_4^+/\text{NO}_3^-$ ratios.

A significant reduction in C/N ratio from $16.5:1 \pm 0.4$ to $14.8:1 \pm 0.3$ within the control windrow ($p < 0.05$) over 8 weeks active composting, indicated a relatively higher loss of carbon than nitrogen (mean C loss = 51.8% , mean N loss = 46.4%). No significant change in C/N ratio was observed in the FT windrow, reflecting similar relative losses of carbon and nitrogen (mean C loss = 55.3% , mean N loss = 55.5%).

4.3.6 Other macro-nutrients (P, K, Mg) including water soluble forms

Table 25 – Phosphorus (P), potassium (K) and magnesium (Mg) contents

Windrow	Sample (week)	Total		Water soluble			Nutrient ratios			
		P %	K %	P mg kg ⁻¹	K mg kg ⁻¹	Mg mg kg ⁻¹	C/P :1	C/K :1	N/P :1	N/K :1
Fresh	0	0.54±0.02	0.33±0.01	296±33	1277±40	360±11	83±1	134±4	5.0±0.2	8.1±0.4
BA	2	0.64±0.01	0.41±0.01	567±65	1693±75	96±12	61±2	94±27	2.7±0.1	4.2±0.2
	4	0.75±0.02	0.46±0.01	489±38	1876±30	77±3	47±2	7±2	2.4±0.1	3.9±0.2
	6	0.73±0.03	0.45±0.02	309±22	1741±48	111±5	47±2	76±4	2.4±0.2	4.0±0.3
	8	0.83±0.01	0.52±0.01	251±2	1531±85	63±8	37±1	59±2	2.3±0.1	3.6±0.1
	12	0.81±0.02	0.45±0.01	174±5	1480±33	71±14	36±1	65±2	2.6±0.1	4.8±0.2
FT	8	0.78±0.01	0.49±0.02	582±70	2417±150	160±32	46±1	74±3	2.8±0.1	4.5±0.3
Control	8	0.82±0.01	0.49±0.01	431±58	2492±50	217±14	45±1	75±2	3.0±0.1	5.1±0.2

Of the initial level of total P in the fresh WPS, only 5.5% was present in water soluble (available) form (0.03%DM), whereas 38.7% of the level of total K was present in water soluble form (0.13%DM).

During the first 2 weeks of active composting there was a significant ($p<0.001$) reduction in total P ($22.6\pm0.7\%$ ash) within the BA windrow, equivalent to 57% of total loss in total P over 12 weeks. This coincided with a net increase in water soluble P of $25\pm2.3\%$ ash. A further reduction of 17.2% ash total P occurred during the remainder of the 12 week experimental period ($p<0.001$).

Total P losses within the FT windrow of $20\pm0.5\%$ ash after 8 weeks active composting was significantly ($p<0.001$) higher than the control windrow ($10.9\pm0.3\%$), the losses for both these windows were significantly less ($p<0.001$) than that recorded for the BA windrow at week 8 ($33.3\pm0.8\%$ ash). The FT windrow showed a net increase in water soluble P of $8.8\pm0.8\%$ ash, whereas the control windrow showed a net reduction in water soluble P of $15.3\pm1.6\%$ ash.

During the first 2 weeks of active composting there was a significant ($p<0.001$) reduction in total K ($19.2\pm0.4\%$ ash) within the BA windrow, equivalent to 41% of total loss in total K over 12 weeks. This coincided with a net decrease in water soluble K of $13.6\pm0.4\%$ ash. A further reduction of 27.4% ash total K occurred during the remainder of the 12 week experimental period ($p<0.001$).

Total K losses within the FT windrow of $19.3 \pm 0.6\%$ ash after 8 weeks active composting was significantly ($p < 0.01$) higher than the control windrow ($14.5 \pm 0.3\%$); the losses for both these windrows were significantly less ($p < 0.001$) than that recorded for the BA windrow at week 8 ($32.9 \pm 0.7\%$ ash). Both the FT and control windrows showed net increases in water soluble K; $4.5 \pm 0.2\%$ ash and $13.3 \pm 0.3\%$ ash respectively.

Pearson correlation analysis showed that mean total K losses (data taken from all three experimental windrows) were significantly correlated with mean total P losses (Pearson $r = 0.96$; $R^2 = 92\%$; $p < 0.001$; using 7 x,y pairs).

All nutrient ratios fell significantly ($p < 0.001$) below initial values for all windrows after 8 weeks of composting, showing that relative losses of carbon and nitrogen exceeded losses of P and K within all windrows. In all cases the BA windrow possessed significantly ($p < 0.05$) lower nutrient ratios after 8 weeks active composting than either the FT or control windrows. No significant difference between the FT and control windrow nutrient ratios was observed ($p > 0.05$).

Large decreases in water soluble Mg were observed for all windrows over 8 weeks active composting; $92.5 \pm 8.1\%$ ash (BA), $75.5 \pm 9.5\%$ ash (FT), and $65 \pm 3.6\%$ ash (control). Due to large variation within the data there was no statistically significant difference between these figures ($p > 0.05$). In the BA windrow most (89.7%) of the water soluble Mg reduction occurred after only 2 weeks.

4.3.7 Changes in acidity/alkalinity (pH) and electrical conductivity (EC).

Table 26. – Changes in pH during active composting

Windrow	Sample (week)	pH	H ⁺	EC μS cm ⁻¹
Fresh	0	7.2±0.0	7.3 × 10 ⁻⁸	1350±66
BA	2	7.8±0.2	2.1 × 10 ⁻⁸	1405±78
	4	7.5±0.0	3.0 × 10 ⁻⁸	1403±52
	6	6.8±0.0	1.5 × 10 ⁻⁷	585±6
	8	7.4±0.2	5.0 × 10 ⁻⁸	430±14
	12	6.9±0.0	1.2 × 10 ⁻⁷	430±9
FT	8	7.5±0.1	4.2 × 10 ⁻⁸	2275±127
Control	8	7.0±0.0	1.0 × 10 ⁻⁷	2193±79

A significant increase in pH occurred within the BA windrow after 2 weeks active composting ($p<0.01$), equating to a 3.5-fold decrease in $[H^+]$. However, after 8 weeks composting there was no longer any significant difference in the BA windrow's pH from the WPS's initial value ($P>0.05$). After a further 4 weeks (maturation period), pH within the BA windrow was significantly below that for week 8 ($P<0.05$), equivalent to a 1.6-fold increase in $[H^+]$ - often associated with the onset of nitrification (see discussion).

At week 8, pH levels of the FT and control windrows were not significantly different from the WPS initial level.

EC within the BA windrow did not change significantly until week 6 of active composting, where it showed a mean reduction of $765 \mu S cm^{-1}$ (57%) ($p<0.001$). This coincides with a large reduction in NH_4^+ . Pearson correlation analysis showed a significant correlation between EC and NH_4^+ ions for all samples taken from all experimental windrows during the 12 week experimental period (Pearson $r = 0.986$; $p<0.0001$; $R^2 = 97.2\%$; 28 x,y-pairs). Pearson's correlation analysis of EC against the total concentration of all water soluble nutrients ($NH_4^+ + NO_3^- + P^{3-} + K^+ + Mg^{2+}$), recorded for all samples from all experimental windrows, showed a slightly stronger correlation (Pearson $r = 0.989$; $p<0.0001$; $R^2 = 97.7\%$).

During 8 weeks active composting the FT and control EC level increased significantly ($p<0.001$), $925 \mu S cm^{-1}$ (69%) and $843 \mu S cm^{-1}$ (62%) respectively. This reflects an overall net increase in water soluble ions during composting, mainly in the form of NH_4^+ , supported by the correlation analyses detailed above.

4.4 Discussion

After week 2 of the composting period the moisture content of WPS within the bulking-agent treated (BA) windrow was maintained at a level generally considered not to be inhibitory to composting processes: 50-75% (Miller 1991) and >35% (Stentiford 1996). However, within both the windrow treated by frequent turning (FT) and the control windrow, moisture contents remained above 75% for the whole composting duration, a level considered inhibitory (*ibid* 1991, *ibid* 1996).

Temperatures in excess of 50°C recorded within the BA treated windrow over the first 4 weeks of active composting suggested that exothermic decomposition, associated with aerobic conditions, was underway. Similar temperatures have been associated with optimal respiration rates in thermophilic fungi (Gray *et al.* 1971), and these fall within the optimum range of 45-60°C suggested by Bardos & Lopez-real (1991), <60°C (Poincelot 1975). A steep decline in temperature within the BA treated windrow after week 4 suggested a deceleration in decomposition processes (Golueke & Diaz 1990; Haug 1993).

Temperatures within the FT treated and control windrow remained below 40°C suggesting that anaerobic conditions within the WPS inhibited exothermic decomposition.

Compositional analysis of gases within the experimental windrows confirmed that conditions within the BA treated windrow were aerobic, with oxygen levels >6% during the first four days of composting, rising significantly ($p<0.001$) to >15% by day 7 and remaining >15% for days 9, 18 and 50. Conditions within the FT treated and control windrows were much more anaerobic. Levels of carbon dioxide during early (days 1-4) and later (days 7, 9, 18 and 50) stages of composting were very significantly higher within the FT and control windrows ($p<0.001$), whilst oxygen levels were very significantly lower ($p<0.001$).

Higher windrow temperatures and aerobic conditions within the BA windrow concur with very significantly greater overall TOM losses, and TOM loss rate suggested by non-linear regression modelling of WPS decomposition within the BA windrow ($p<0.001$, TKMCT). The FT windrow showed a small (2%DM), but just significant ($P<0.05$, TKMCT), greater overall loss in TOM, as shown by non-linear modelling and actual losses (%ash).

After 12 weeks composting the final TOM content of the BA windrow composted WPS was 52.4%DM, 63.2%DM for FT treated WPS, and 64.9%DM for the control. These relatively high proportions of organic matter remaining after composting probably reflect the recalcitrant nature of the feedstock. For comparison, figures obtained for other composted materials in other investigations are: 36.9%DM - cattle slurry co-composted with rice hulls for 36 weeks, 8 weeks active composting (cylindrical adiabatic reactor) plus 28 weeks curing

(Genevini *et al.* 1996); and 39.5%DM - a mixture of waste paper sludge, chicken litter and yard waste (8:2:1 by volume) after 27 weeks composting, 7 weeks active composting (aerated static pile) plus 20 weeks maturation (Sesay *et al.* 1997).

However, overall losses in TOM of 74.1%ash obtained for the BA treated WPS, 59.7%ash (FT treated), and 56.6%ash for the control, are broadly comparable to losses observed for other materials (composted using varying composting methods and durations): 60% - food waste and leaves (Michel *et al.* 1996); 52% cattle manure (Tarre *et al.* 1987); 75% - green waste (Frederickson *et al.* 1997); and 55% - a mixture of waste paper sludge, chicken litter and yard waste (Sesay *et al.* 1997).

The losses in TOM observed within the FT and control windrows, despite largely anaerobic conditions, indicate the readily biodegradable nature of some fractions of the WPS.

The initial C/N ratio of the WPS feedstock (16.5:1) was lower than recommended ranges: 30-35:1 (Gray *et al.* 1971); 26-35:1 (Poincelot 1975); 30-50:1 (Taiganides 1977); 20-30:1 (Jansson & Persson 1982); 20-35:1 (Haug 1993). A mean overall loss in total nitrogen of 68.5%ash within the BA treated windrow indicates an excess in available nitrogen compared with carbon.

Nitrogen losses via volatilisation would have been encouraged by high temperatures, high ammonium concentrations and high pH (Bernal *et al.* 1993) during the first 4 weeks of composting, during which 93% of nitrogen losses in the BA windrow occurred. Gas analysis also identified high concentrations of ammonia during the early days of composting.

Despite lightly covering the windrows during periods of excessive rainfall, some leaching of N, P and K may have occurred. Significantly greater losses were observed for the BA treated windrow ($p < 0.01$) and this may suggest that the increased porosity created by the bulking agent may have aided the percolation of water through the pile. Another probable cause of N, P and K reductions was their absorption, in water soluble form, by the bulking-agent.

However, nitrogen losses within the FT (55.5%ash) and control windrow (46.4%ash) were significantly lower despite their consistently higher moisture contents. This may suggest that nitrogen loss was largely through volatilisation, increased within the BA windrow through greater ventilation. This is further supported by a significantly greater nitrogen loss through volatilisation observed for the FT windrow than the control windrow ($p < 0.01$), encouraged by more frequent turning (De Bertoldi *et al.* 1982a).

Despite large losses in nitrogen through volatilisation during the early stages of composting, the C/N ratio of the WPS had been significantly reduced by the BA treated windrow composting to 13.5:1, falling within suggested levels for stabilised materials: 5-20:1 (Hitai

1983), $\leq 30:1$ (ORCA 1992). However, final C/N ratios of the FT composted and control WPS of 16.6:1 and 14.8:1, respectively, also fall into these ranges, when they have clearly not undergone the same level of stabilisation. N/P ratios of between 5:1 and 20:1 for microbial cells (Alexander 1977) suggest a composting feedstock should contain phosphorous at levels between 5 to 20% of the concentration of nitrogen. Gray *et al.* (1971) suggest a C/P ratio of 75:1 to 150:1. The N/P and C/P ratios of the fresh WPS were 5:1 and 83:1, suggesting P content was not a limiting factor.

Large levels of water extractable K within the BA treated WPS after 12 weeks composting suggest that K was present in excess of microbial requirements, *i.e.* immobilisation did not exceed mineralisation (Swift *et al.* 1979).

The production of nitrate within the BA composted WPS at weeks 8 and 12 suggests favourable composting conditions (Miller 1992) and the onset of maturation (Finstein & Miller 1985). A low final level of NH_4^+ (0.0006%DM) within the BA treated WPS after 12 weeks also indicates a stabilised material, falling below suggested limits of 0.04%DM (Forster *et al.* 1993, and Bernal *et al.* 1998) and 0.005%DM (Avnimelech *et al.* 1996). A $\text{NH}_4^+/\text{NO}_3^-$ ratio of 0.021:1 also suggested the maturity and stability of the final BA treated WPS compost, falling below a suggested limit of 0.16:1 (Bernal *et al.* 1998).

No nitrification was observed within the FT treated and control WPS and very high NH_4^+ levels were retained. This in turn led to very high $\text{NH}_4^+/\text{NO}_3^-$ ratios after 8 weeks. Nitrification is an aerobic process, and may have been inhibited by anaerobic conditions within the FT and control windrows created by a lack of physical structure and high moisture contents. This would also explain the large build up of NH_4^+ , which would require ventilation/aeration for volatilisation as NH_3 .

The stability of WPS composted within the BA treated windrow after 8 weeks was further supported by 95% completion of total TOM losses. Gas analysis also revealed that, at day 50 (7 weeks), the CO_2 and O_2 concentrations of gas within the BA windrow was no longer significantly different from ambient air.

4.5 *Conclusions*

The findings of this study suggest that the decomposition and stabilisation rates of commingled WPS were significantly improved by the addition of a bulking agent in a ratio of 1:1 by volume, prior to mechanically-turned windrow composting. Good rates and levels of decomposition were achieved using a relatively infrequent turning rate (once per week).

Without the use of a bulking agent successful composting of the relatively wet WPS feedstock was not achieved. Although an increased turning regime alone did have a slight effect on overall TOM decomposition and nitrogen loss, the WPS did not fully decompose or stabilise due to rapid build up of anaerobic conditions.

Within 8 weeks of composting using a bulking-agent, 95% of TOM decomposition was completed, with the onset of nitrification indicating a mature product. The final composted WPS retained high levels of nutrients N, P, K, possessed correspondingly low carbon/macro-nutrient ratios, and its nitrate content continued to rise during maturation. A high final TOM content could make the final composted WPS a useful soil amendment, or a potential component of plant growth media (investigated in Chapter 6).

The use of a bulking agent when composting these types of commingled sludges would appear to be highly justified. A detailed comparison of the process of windrow composting commingled sludge with the process of vermicomposting similar sludge will be made in subsequent Chapters. Comparison of the composted products arising from both these processes will also be made.

5. Large scale vermicomposting of a commingled primary and secondary waste paper sludge using a modular batch vermicomposting system.

5.1 Introduction

As with studies into the windrow composting of WPS, studies into the vermicomposting of WPS have shown that many waste paper pulps require nutritional amendment to induce adequate levels of microbial activity and earthworm nutrition during vermicomposting. Butt (1993) highlighted that WPS types can vary considerably, and may require amendment prior to worm processing. Elvira *et al.* (1997) vermicomposted WPS after its amendment with wastes that contained high levels of nitrogen (animal manures). Unfortunately such amendments resulted in high levels of earthworm mortality (comparable to the primary and secondary WPS mixture investigated in Appendix 2).

However, some investigations into WPS vermicomposting have shown certain waste paper sludges can be successfully processed without amendment (Fayolle *et al.* 1997). Edwards & Burrows (1988) state that WPS is one of the most suitable wastes for vermicomposting, but the precise composition of the WPS used was not given.

Building on the research detailed in Chapter 3 this study investigates the vermicomposting of a commingled primary and secondary waste paper sludge, considered not to require any nutritional amendment. This enabled the investigation, in isolation, of the effect of two different medium scale vermicomposting systems on the WPS feedstock. As for the windrow composting experiments, where it was necessary to determine the most effective composting regime, it was considered important to attempt to optimise the method of vermicomposting. This is particularly important in the context of comparing the two processes and the resulting composted products in subsequent Chapters.

Two medium scale vermicomposting systems were therefore developed and set up. These systems were based on existing approaches to vermicomposting (Edwards 1998). The two systems were similar to batch type systems (single-batch vermicomposting) and continuous flow systems (multiple-batch vermicomposting). In the batch system worms are in contact with the substrate for several weeks, whereas with continuous flow systems the worms, having processed the substrate move away into freshly applied waste. In addition to comparing the two vermicomposting systems, one system (single-batch vermicomposting) was set up mainly to investigate the effect of earthworms on a single quantity of WPS over 8 weeks (research question 1 below); and to produce a sufficient quantity of vermicompost for plant growth

trials (Chapter 6). The second system (multiple-batch vermicomposting) was mainly set up to produce a sufficient quantity of discrete WPS earthworm casts, which were then incubated without earthworms to assess their stability and subsequent physicochemical changes.

Research hypotheses:

1. The addition of earthworms to WPS will accelerate the decomposition and stabilisation of WPS, through increased aeration, decreased particle size (increased surface area) of WPS, and increased microbial activity of WPS during vermicomposting.
2. The process of WPS ingestion and egestion by worms as casts will partially stabilise WPS, a process which will continue during compost maturation in the absence of earthworms.

Additional research questions:

1. How do earthworms affect the decomposition and stabilisation of unamended WPS during vermicomposting?
2. Is vermi-processed (ingested and egested) WPS (casts) a stabilised material, and how do the physicochemical characteristics of WPS casts change in the absence of earthworms?

5.2 Materials and methods

5.2.1 Feedstock preparation

A feedstock of commingled primary and secondary WPS was obtained from a packaging manufacturer (SCA Packaging Ltd.). The WPS consisted of a mixture of primary clarifier residues and secondary (activated) sludge belt pressed into a sludge cake with a solids content of 20–25%. The fresh WPS delivered to the experimental site was randomly sampled to allow physicochemical analyses, and sufficient sludge was removed for the medium-scale vermicomposting experiments. The remainder was divided into three equal fractions, each containing approximately 5 tonnes of material, and this was used for the windrow composting experiments.

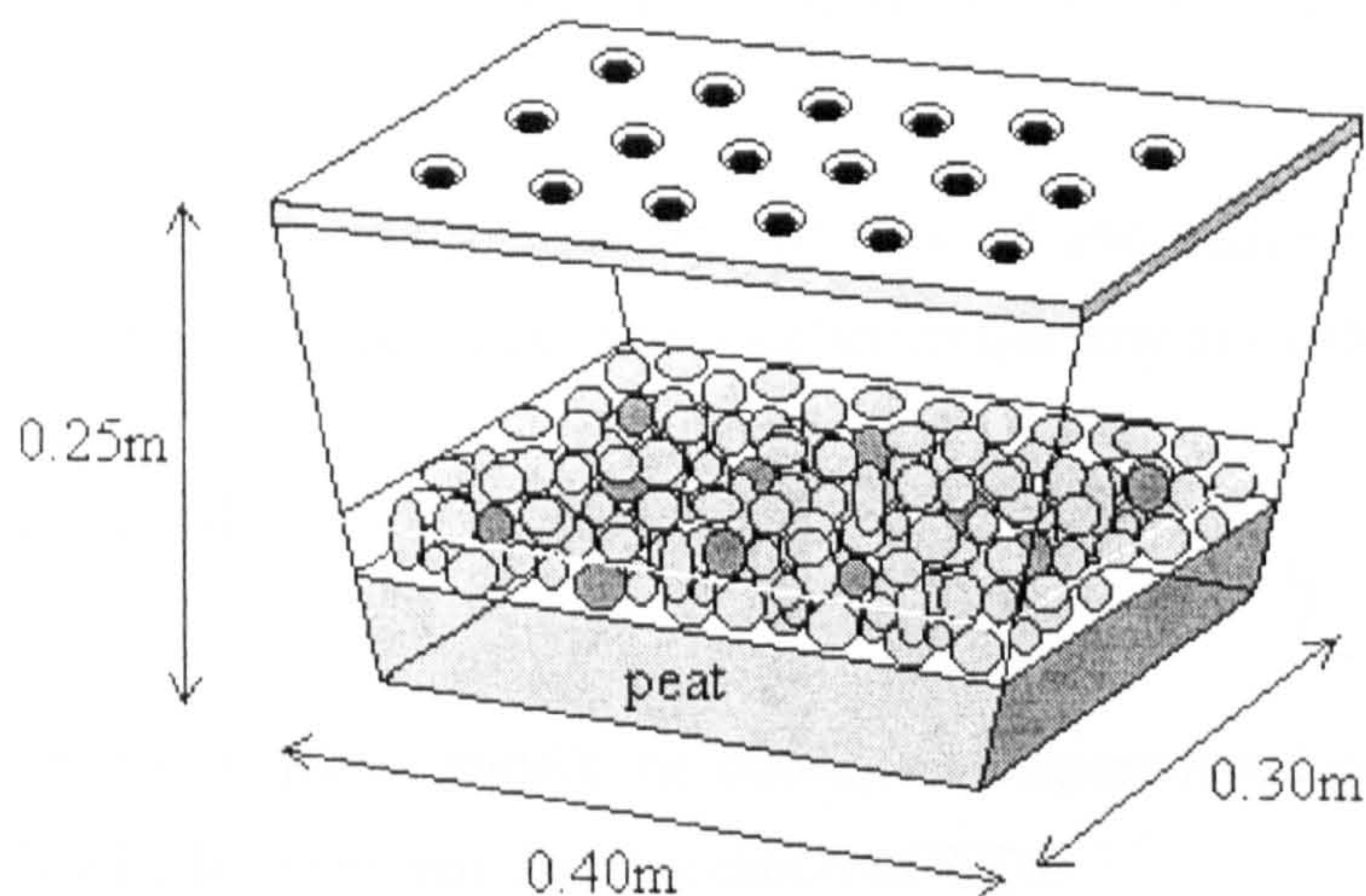
Only fresh WPS was added to vermicomposting units, as ageing produced increased levels of ammonia, known to be highly toxic to earthworms.

5.2.2 Vermicomposting units (VCUs)

Vermicomposting was carried out using individual, modular vermicomposting units (VCUs) (Figure 30). A total of 40 VCUs were constructed (Appendix 3); 32 active (containing earthworms) and 8 inactive (without worms) control VCUs. The single batch and multiple batch experiments were each allocated 16 active and 4 inactive VCUs.

Each VCU consisted of a plastic trough (length 0.40 m, width 0.30 m, height 0.25 m) partially filled with 5 litres of moss-peat bedding material, and then covered with 6 kg of beach pebbles (diam. 2–3 cm) to prevent the mixing of WPS and peat during vermicomposting. Active vermicomposting units were stocked with 500g clitellate *Dendrobaena veneta* (a suitable vermicomposting earthworm species) with a mean individual wet mass 1.4 ± 0.1 g; adult worms were used to achieve a maximum processing rate of WPS (Chapter 3; Edwards 1988). This is equivalent to a stocking density of 4 kg worms m^{-2} VCU area ($20 \text{ kg } m^{-3}$ VCU volume). This density was selected to concur with findings using similar epigeic earthworm species; 7-11kg m^{-2} obtained for *Eisenia andrei* on pig manure (Reeh 1992) and 2kg m^{-2} observed for *Eisenia fetida* (Hartenstein 1981), as well as carrying capacities observed during earthworm growth experiments (Chapter 3). Highest levels of earthworm survival, total worm mass production, and WPS processing were obtained using the highest densities investigated (8 to 16 worms per 100g-WPS 2-weeks $^{-1}$). This equated to a initial stocking density of 0.65 to 0.13 kg-worm kg-WPS $^{-1}$. Multiplying this by 6 kg of WPS used in the single-batch system a total worm mass of 390 to 790g, respectively. It was decided that a stocking density at the mid point of this range would provide optimal WPS processing, while maintaining adequate earthworm biomass.

Figure 30 – vermicomposting unit



5.2.3 Vermicomposting

Single-batch vermicomposting

The main aim of this experiment was to investigate the effect of earthworms on the decomposition and stabilisation of WPS over 8 weeks active vermicomposting plus 4 weeks maturation. It was also designed to produce enough vermicompost for plant growth trials (Chapter 6). Each of the 16 active VCUs was randomly assigned to one of four replicates (ie 4 VCUs per replicate). To each active VCU, 6 kg (~15 litres) of fresh WPS was added; giving a stocking ratio of 12:1 WPS to worms.

Inactive VCUs, without worms, ($n=4$) containing the same quantity of bedding material and fresh WPS were used as controls to monitor the decomposition and physicochemical changes within WPS in the absence of earthworms.

WPS was actively vermicomposted for 8 weeks, after which all casts and unprocessed WPS were removed from VCUs and matured in the open air (inside the heated polythene tunnel) for a further 4 weeks.

Multiple-batch vermicomposting

The main aim of the multiple batch experiment was to investigate the effect of earthworm processing (ingestion of WPS and egestion as casts) on the physicochemical characteristics of WPS, including subsequent decomposition during maturation. Active VCUs ($n=16$) had 2 kg (5 litres) of fresh WPS added to them to produce worm casts over two weeks vermicomposting.

After 2 weeks vermicomposting all WPS was removed from each active replicate ($n=4$) consisting of a set of 4 VCUs and casts were hand-sorted from unprocessed WPS. The casts were separated into equal portions; one portion was stored for analysis, and the other was

incubated without worms for a further 10 weeks to give a total of 12 weeks processing. Similarly, WPS from each VCU without worms ($n=4$) was extracted after two weeks and matured as before.

This vermicomposting process and subsequent cast removal and maturation was repeated a further three times to ensure the production of a sufficient quantity of earthworm casts.

5.2.4 Process Control

Moisture content

No additional water was required initially or during the experimental period. VCU's were covered with ventilated lids to prevent the desiccation of WPS.

VCU temperature

VCU temperatures were maintained at $20.8 \pm 0.7^\circ\text{C}$ inside a thermally insulated polythene-tunnel, using thermostatically controlled under soil cable heating. Temperatures were monitored using $8 \times 1 \text{ m}$ temperature probes connected to a digital data-logger (IceSpy[®], © Silvertree Engineering Limited). One probe was placed in each of 8 control VCUs, at the interface between WPS and beach-pebble layers. Data-loggers recorded the temperature of each probe at 10 minute intervals, and average weekly temperatures were calculated from daily readings.

5.2.5 Process monitoring

Earthworm population

Total and individual earthworm mass was weighed before and after 8 weeks of single-batch vermicomposting to monitor earthworm growth. Cocoons, and hatchlings were counted to measure net reproduction over the 8 week experimental period.

The final vermicompost was hand-sorted and all adult worms, cocoons and hatchlings were separated, counted and weighed. Cocoon viability was measured by incubating cocoons in petri-dishes on saturated filter paper at $20\text{-}21^\circ\text{C}$ for 6 weeks.

These parameters were used to determine whether a stable earthworm population was maintained during vermicomposting, and thus adequate vermicomposting had been achieved.

Sampling regime

Samples of WPS and WPS plus casts were collected at weeks 0 (fresh) 1, 2, 4, 6, 8 and 12 (*i.e.* after 4 weeks maturation) for both single and multiple batch experiments.

In the single batch experiment (and the week 1 sample for the multiple batch experiment), approximately 0.4 kg samples were taken from each of four VCUs per replicate, and aggregated into one 1.6 kg sample per replicate ($n=4$) at each sampling period.

The multiple batch experiment was run 4 times. For each run the sample for week 1 of each run consisted of a mixture of casts and unprocessed (non-ingested) WPS; whereas week 2 samples consisted purely of WPS casts. Samples for weeks 4, 6, 8, and 12 consisted of those casts that had been matured without earthworms.

All samples were kept under frozen storage in hermetically sealed plastic bags until analysis.

Physicochemical analyses

Moisture content

All samples were analysed for moisture content (% wet mass). Moisture content was estimated gravimetrically as the loss in mass of a 100g portion of sample dried in a fan assisted desiccating oven at 105 °C for 24 hours.

Total organic matter and carbon contents

Total organic matter content (TOM) was determined for all samples. A 100g portion of each sample was dried at 105°C for 24 hours, and milled to a <1 mm particle size using a Messerm-125H knife-mill. Portions (approximately 5g) of the prepared samples were placed in ceramic crucibles and ignited in a muffle furnace. TOM was calculated as the loss in mass (volatile solids) after ignition at 550°C for 2 hours (FCQAO, 1994).

Total organic carbon (TOC) content was calculated to be 55% of the TOM (Adams *et al.*, 1951); this falls within the range of 45 to 60% recorded for some recycled WPSs (Bellamy *et al.* 1995).

*Water extractable ions**

Water soluble ammonium (NH_4^+), nitrate (NO_3^-), phosphorous (PO_4), potassium (K^+) and magnesium (Mg^{2+}); pH and electrical conductivity (EC) were determined for fresh feedstock and samples taken after 2, 4, 6, and 12 weeks.

* Analyses were conducted by 'Levington Agriculture', Levington Park, Ipswich, Suffolk, UK; using standard ADAS procedures.

Total (Kjeldahl) nitrogen (N), total phosphorus (P) and total potassium (K) contents were determined for fresh feedstock and samples taken after 2, 4, 6, and 12 weeks.

Mathematical and Statistical analysis

Relative nutrient losses (correcting for dry mass losses)

Nutrient losses were corrected for a concentration effect due to the reduction in organic matter during vermicomposting. Initial ash content (%DM) was used as a baseline, and was assumed to remain at a constant total mass (Stentiford & Pereira Neta 1985).

Using the theory of ash conservation, dry mass (DM) contents can be calculated using the following equation:

$$DM_t = TOM_t + ash_0 \quad (1)$$

TOM = total organic matter content after vermicomposting duration (t); ash₀ = initial ash content (adapted from Genevini *et al.* 1996).

Thus nutrient losses may be corrected for loss of DM using the conservation of ash:

$$X\%_{ASH} = \left\{ 1 - \left(\frac{\%X_t \times \%ash_0}{\%X_0 \times \%ash_t} \right) \right\} \times 100 \quad (2)$$

X = nutrient; %ash₀ = initial ash content (%DM); %X₀ = initial nutrient content (%DM); %ash_t = final ash content (%DM) after vermicomposting duration (t); %X_t = final nutrient content (%DM) after vermicomposting duration (t) (adapted from Bernal *et al.* 1996).

Non-linear regression of TOM with vermicomposting duration

Curves were fitted to TOM (%DM) data against vermicomposting duration, using a one-phase exponential decay model:

$$TOM (\%DM) = a \times e^{(-\lambda x)} + c \quad (3)$$

a = total loss in TOM (%DM) (initial TOM – c); c = final value of TOM (%DM);

λ = decay constant; Half-life (T) = (ln2)/λ.

Curves were fitted using SPSS[®] statistical analysis software package (© SPSS Inc.), using the Levenberg–Marquardt method of estimation (sum-of-squares minimisation).

Though usually associated with radioactive decay, the one-phase decay model provided scientific assumptions felt appropriate for modelling organic matter decomposition, *i.e.*, a decelerating rate of decay before reaching an essentially constant final value. The same model

was used by Vincelas-Akpa & Loquet (1997) to fit curves to losses of organic carbon and cellulose during the composting and vermicomposting of lignocellulosic (maple) waste.

The model was used to estimate values of total loss of TOM (%DM); final value of TOM (%DM); and the rate at which these losses occurred (λ & T).

Statistical differences between decay curves were calculated from the means and asymptotic standard errors (se) of variables (a, λ , and c). A one-way analysis of variance (ANOVA) was used to find statistical differences between the variables of different experimental VCU; statistical differences between specific pairs of VCUs were analysed using Tukey–Kramer multiple comparison post ANOVA tests.

Statistical analysis of acidity/alkalinity

Statistical differences in acidity/alkalinity were calculated using $[H^+]$; pH was converted into $[H^+]$ using equation 4.

$$[H^+] = 10^{-pH} \quad (4)$$

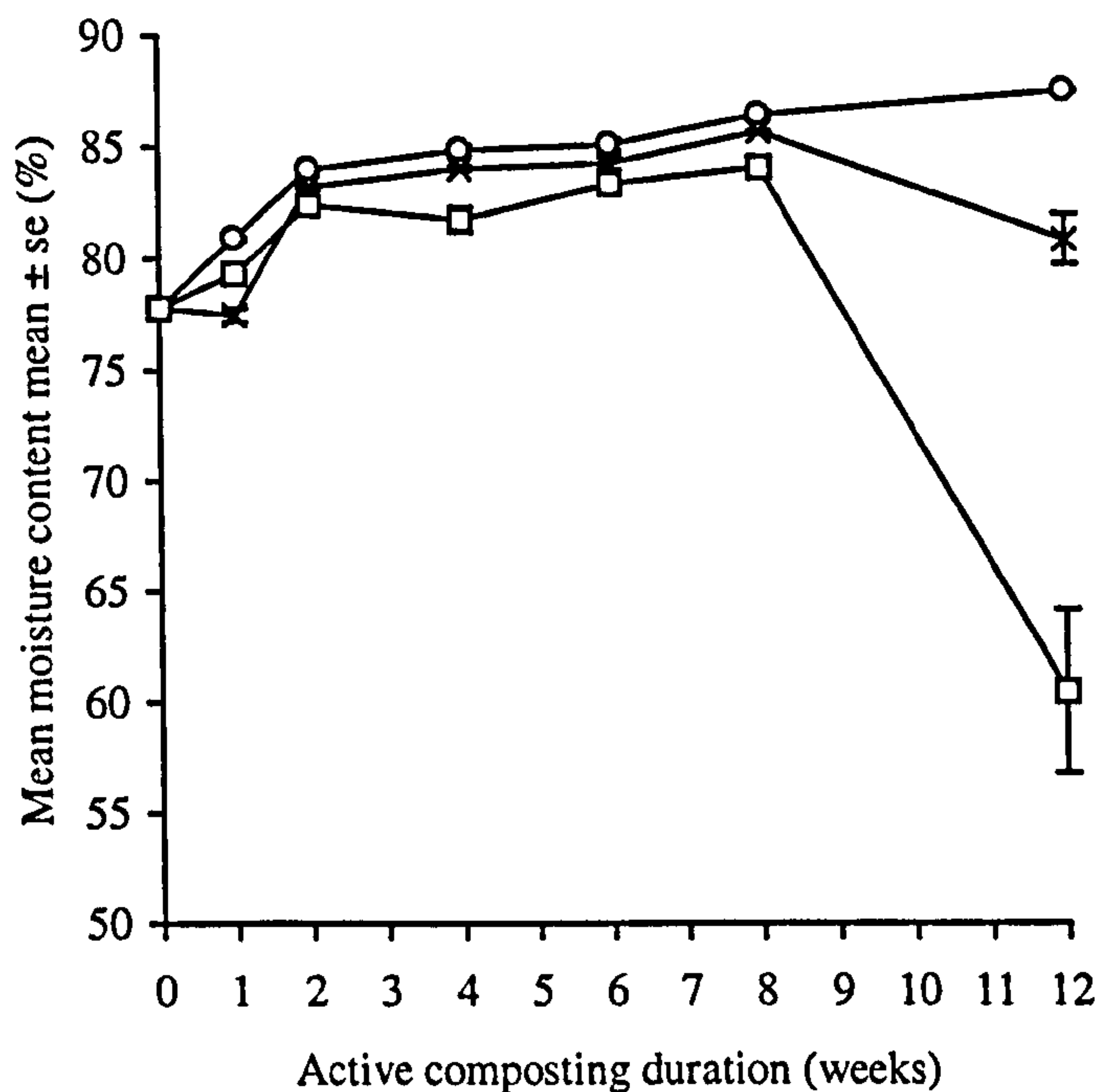
General statistical analysis

Statistical analyses of differences between means were performed using standard two-tailed Student t-tests, assuming equal standard deviations of means. Modified t-tests or other statistical methods utilised are stated next to p-values.

5.3 Results

5.3.1 Moisture

Figure 31 – Changes in moisture content; Single-batch VCUs [—□—]; Multi-batch VCUs [—×—]; Control VCUs [—○—].



The initial moisture content of the WPS (without addition of water) was 77.8%, within the range of 60-90%, suggested for vermicomposting (Edwards 1988). During the first 2 weeks of the experimental period moisture levels rose significantly ($p < 0.001$) within all VCUs by between 4.7 and 6.3% wet mass (with no significant difference between VCUs).

During weeks 4–8 of the experimental period, mean moisture contents within the single-batch (SB), multiple-batch (MB) and control VCUs varied little; fluctuating between 81.8–84.1%, 84.9–87.6%, and 84.1–85.5%, respectively.

Final moisture contents (week 12, after 4 weeks maturation) of, $60.5 \pm 3.7\%$ (single-batch VCUs), $87.6 \pm 0.5\%$ (multiple-batch VCUs), and $80.9 \pm 1.1\%$ (control VCUs) were all significantly different ($p < 0.001$, post ANOVA Tukey-Kramer multiple comparisons test)

5.3.2 Total organic matter

Table 27 – Loss of TOM (%ash*); mean±se.

Week	SB	MB	Control
1	7.7±0.3 ^a	24.4±0.3 ^b	7.8±0.3 ^a
2	25.4±1.0 ^a	37.4±0.9 ^b	21.8±0.9 ^c
4	37.8±0.9 ^a	45.6±0.6 ^b	33.7±0.8 ^c
6	45.3±1.1 ^a	46.7±2.3 ^a	41.2±1.3 ^b
8	52.7±2.8 ^a	49.8±0.3 ^a	39.6±0.3 ^b
12	62.3±0.5 ^a	54.2±0.3 ^b	50.6±0.5 ^c

* for calculations see section 0

^{abc} different letters indicate significant differences (p<0.05, post ANOVA Tukey–Kramer multiple comparisons test).

Table 28 – Non–linear regression of TOM (%DM) vs. vermicomposting duration; mean±se (week 0 to 8).

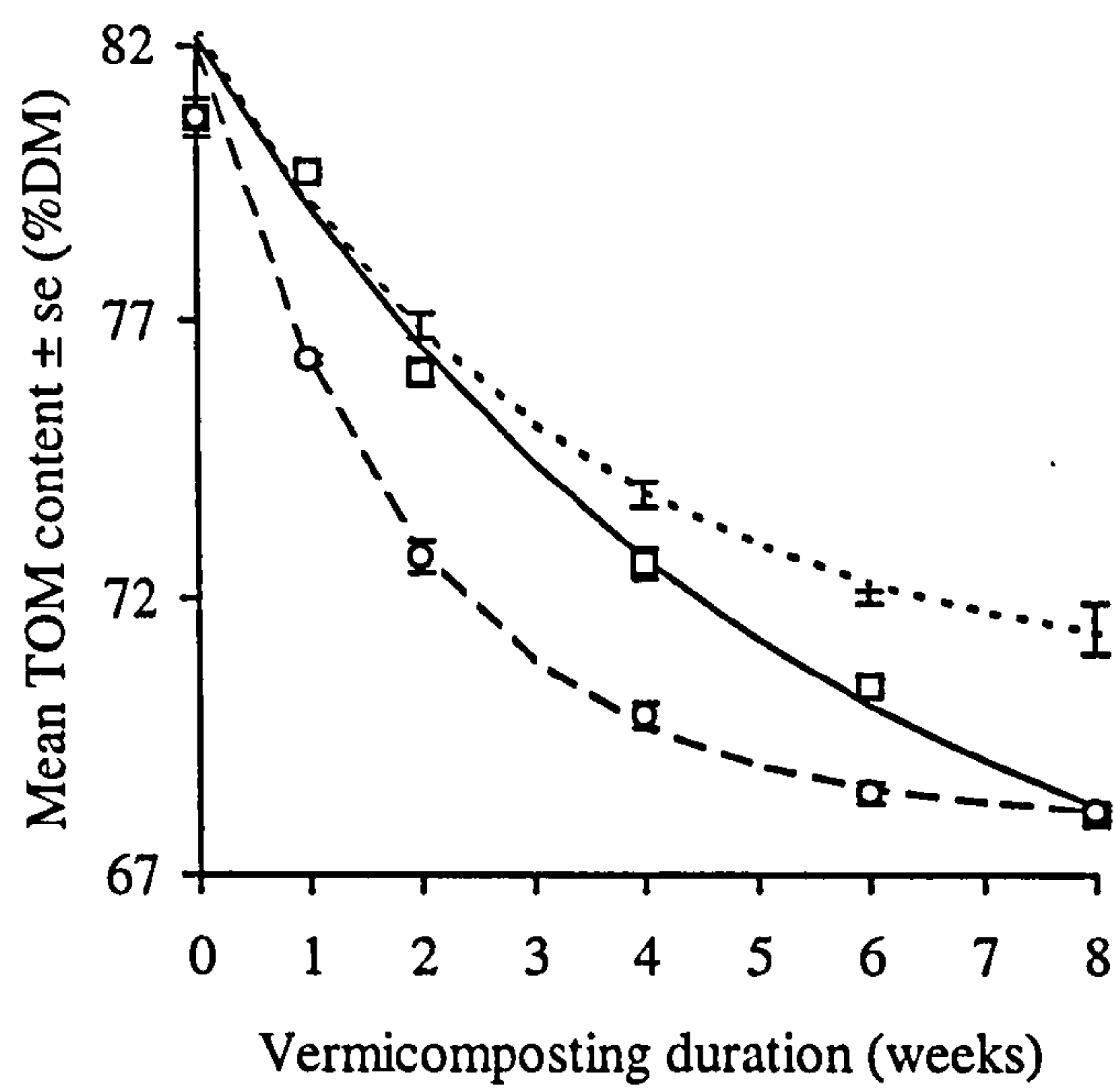
Variable	SB	MB	Control
Total TOM loss (a)	17.92±1.09 ^a	13.99±0.23 ^b	11.99±0.60 ^c
Final TOM (c)	64.23±1.18 ^a	67.95±0.18 ^b	70.27±0.64 ^c
Decay rate (λ)	0.187±0.023 ^a	0.521±0.023 ^b	0.297±0.04 ^b
Half–life (T) weeks	3.71±0.46 ^a	1.33±0.06 ^b	2.33±0.32 ^c
50% of total TOM loss rate (TOM %DM wk ^{–1})	2.42±0.23 ^a	5.26±0.16 ^b	2.57±0.24 ^a
Measures of goodness of fit			
R ²	98.9%	99.5%	97.2%
S _{x,y} (TOM %DM)	0.59	0.38	0.70

^{abc} different letters indicate significant differences (p<0.05, Tukey–Kramer multiple comparisons test).

R² = fraction of total variance in experimental data explained by the model.

S_{x,y} = standard deviation of points from the model line–of–best–fit. (Motulsky 1996)

Figure 32– Changes in TOM content (%DM) with vermicomposting duration; SB VCU [□, best fit = —]; MB VCU [×, line of best fit - - -]; control VCU [○, line of best fit = - - -].



Non-linear regression analysis using a one-phase-exponential decay model provided a good fit for TOM %DM data (Table 28); with >97% of data variance explained by the model, and a standard deviation of data from the model of ≤0.1%DM TOM.

The model shows that the WPS processed in the SB VCU, gave a significantly greater loss in total TOM %DM and lower final TOM %DM, than both the MB and control VCUs ($p < 0.05$). Mean differences in final TOM %DM predicted by the model between the SB and MB VCUs and the control were 3.72 and 6.04 %DM respectively.

Total losses in TOM %DM, and final TOM %DM levels predicted by the model, were also significantly different ($p < 0.05$) between the MB and control VCUs (see Table 28).

The rate at which these total TOM losses occurred, *i.e.* the rate at which the decay curves reach a plateau, also differed significantly between VCUs ($p < 0.0001$, ANOVA). Within the MB VCUs, losses of TOM %DM decelerated most rapidly, with a significantly shorter half-life, than either the SB or control VCUs ($p < 0.01$, post ANOVA Tukey-Kramer multiple comparisons test). The MB achieved 50% of its total TOM losses (7.0%DM) in 1.33 weeks, giving a rate of 5.26 %DM TOM week⁻¹, which is significantly higher than achieved for either the SB or control VCUs ($p < 0.001$, post ANOVA Tukey-Kramer multiple comparisons test). The 50% total TOM loss rates of the SB and control VCUs do not differ significantly, indicating similar rates of decay (TOM %DM wk⁻¹) were achieved during the first half of their

respective total TOM losses. However, due to its significantly lower final TOM content, TOM losses within the SB VCU must extend over a longer period, indicated by a significantly longer half-life ($p < 0.05$, unpaired Student *t*-test, equal standard deviations).

Losses of TOM (%ash) (Table 27) support the main findings of the non-linear analysis, showing significantly greater losses within the MB VCU for weeks 1 to 4, with TOM (%ash) losses no longer significantly different between the MB and SB VCUs for weeks 6 and 8 ($p > 0.05$). Between weeks 2 and 8 both the SB and MB VCUs show greater reductions in TOM (%ash) than the control VCUs ($p < 0.01$, post ANOVA Tukey-Kramer multiple comparisons test).

From Table 1 it can be seen that after 8 weeks both SB and MB processed WPS show no significant difference in their TOM (%ash) loss; both being significantly greater than the control. This is despite the prediction of the non-linear model that there should be a significant difference in total TOM losses between the SB and MB WPS.

However, after a further 4 week maturation period (in the absence of worms) the SB VCU processed WPS showed a further significant mean reduction in TOM (%ash) ($p < 0.001$) of 9.6%ash (5.1%DM). The final overall loss in TOM was $62.3 \pm 1.3\%$ ash, giving a final TOM value of $61.6 \pm 0.3\%$ DM. The total TOM loss (%ash) observed in the SB VCUs was significantly higher than observed for either the MB vermicomposted or control WPS, as predicted by the model.

The MB and control VCUs also showed significant losses in TOM over the 4 week maturation period ($P < 0.001$) of 4.4%ash (2.0%DM) and 11.0%ash (4.2%DM), respectively. This gave final TOM values of 66.1 ± 0.1 and $67.8 \pm 1.1\%$ DM, respectively, which, again predicted by the model, were significantly different.

Final TOM (%DM) values (week 12, after maturation) of the MB, and control WPS, were significantly lower than final values predicted by the non-linear model ($p < 0.05$, Welsh's corrected Student *t*-test). Mean differences between final TOM %DM values predicted by the model and actual values were 1.85%DM (MB WPS) and 2.5%DM (control WPS). No significant difference was observed between predicted and actual values for the SB vermicomposted WPS ($p > 0.05$, Welsh's corrected Student *t*-test), despite a mean difference of 2.6%DM. This was probably due to large variation of the final TOM%DM variable within the model.

The MB VCU-produced casts (week 2) showed a significantly greater reduction in TOM (%ash) than the WPS plus casts taken from the SB or control VCUs. After week 2, the MB casts continued to show significant reductions in TOM (%ash) over time, in the absence of

earthworms; albeit at a reduced rate. Approximately 69% of the total TOM losses observed for MB processed WPS over 12 weeks, occurred during the first 2 weeks (in the presence of worms) during cast formation.

5.3.3 Nitrogen content

Table 29 – Change in the various forms of nitrogen content *

VCU	Sample (week)	TKN %	N loss %ash	TOC %	C/N ratio	Water soluble		NH ₄ ⁺ /NO ₃ ⁻ ratio
						NH ₄ ⁺ mg/kg	NO ₃ ⁻ mg/kg	
					:1			:1
Fresh	0	2.70±0.06	n/a	44.6±0.1	16.5±0.5	686±78	19±1	36±6
SB	2	3.58±0.09	-5.4±0.1	41.8±0.1	11.7±0.3	2147±150	26±3	82±17
	4	3.42±0.07	12.2±0.2	39.9±0.2	11.7±0.3	4167±701	28±5	148±51
	6	2.54±0.25	40.3±2.4	38.5±0.4	15.1±1.8	1185±77	45±14	26±10
	8	2.78±0.11	41.3±1.9	36.7±0.8	13.2±0.8	417±39	2656±305	0±0
	12	2.47±0.04	54.7±1.0	33.9±0.2	13.7±0.3	357±9	6111±230	0±0
MB	2	3.16±0.07	18.5±0.4	40.0±0.2	12.7±0.3	2550±126	21±2	124±16
	4	2.99±0.19	30.1±1.3	38.4±0.1	12.8±0.9	1352±106	20±1	67±8
	6	2.97±0.08	33.7±0.8	37.7±0.1	12.7±0.4	1086±47	8±2	138±48
	8	3.21±0.12	29.1±0.9	37.5±0.1	11.7±0.5	1043±5	19±1	55±4
	12	2.57±0.08	46.6±1.2	36.4±0.1	14.1±0.5	1115±72	6±1	174±39
control	2	4.09±0.14	-25.5±0.7	42.3±0.1	10.3±0.4	3004±312	20±3	154±40
	4	3.80±0.11	-2.2±0.1	40.6±0.1	10.7±0.4	8100±379	25±2	325±37
	6	3.64±0.11	10.2±0.3	39.3±0.3	10.8±0.4	2761±181	114±82	24±19
	8	3.68±0.06	7.5±1.5	39.6±0.1	10.8±0.2	1117±86	23±1	49±5
	12	2.47±0.32	46.2±3.4	37.3±0.1	15.1±2.3	367±92	594±186	0±0

*All figures expressed in terms of dry mass where applicable.

During 8 weeks active vermicomposting, the single-batch (SB) VCU composted WPS showed a loss of 41.3±1.9%ash from its initial level of TKN (p<0.01), significantly higher than either the MB vermicomposted or control WPS. This fell further to a total loss of 54.7%ash after 4 weeks maturation (week 12). The majority of TKN loss within the SB vermicomposted WPS (74%) had occurred by week 6. The SB vermicomposted WPS final TKN loss (%ash) was significantly higher than observed for the MB vermicomposted WPS (p<0.01).

TKN loss within the MB vermicomposted WPS after 2 weeks (earthworm casts) was 18.5%ash. This was significantly higher than observed for either the SB or control WPS at the same stage (p<0.001). In fact, within the SB vermicomposted and control WPS there appeared to be an increase in nitrogen after 2 weeks. TKN loss within the MB produced casts continued in the absence of earthworms, reaching a final total of 46.6%ash (week 12). Again the majority of this TKN loss (72%) had occurred by week 6.

The control showed an initial increase in TKN of 25% after 2 weeks (P<0.001). Although, after 12 weeks within the control WPS, a loss of 46.2%ash TKN was observed. This was largely due to a 38.8%ash loss in initial TKN over the 4 week maturation period (week 8 to 12), equivalent to 84% of the total TKN loss.

A significant reduction in C/N ratio by week 2 was observed for all VCUs (p<0.001).

The control VCUs showed the greatest reduction ($p < 0.05$), mainly due to an increase in TKN. The C/N ratio of the control WPS remained unchanged ($p > 0.05$) until week 12, where it returned to a figure no longer significantly different from its initial value ($p > 0.05$), this was due to a substantial TKN loss during maturation (see above).

During weeks 2 to 12 the reduction in C/N ratios within the MB vermicomposted WPS was not statistically significant, but by week 12 the final C/N ratio was significantly below the level for fresh WPS ($p < 0.05$).

Within the SB vermicomposted WPS the C/N ratio rose from significantly lower values at weeks 2 and 4 ($p < 0.05$) to a value that was no longer significantly different from fresh WPS at week 6 ($p > 0.05$). However, by weeks 8 and 12 the SB vermicomposted WPS C/N ratio had again dropped to levels significantly below the initial value for fresh WPS ($p < 0.01$).

Large net formation of ammonium (NH_4^+) was observed within all VCUs. The WPS within the control VCUs showed the greatest increases of $260 \pm 28\%$ and $757 \pm 61\%$ over initial values at weeks 2 and 4; coinciding with observed increases in TKN. The SB vermicomposted WPS showed lower increases of $148 \pm 14\%$ and $321 \pm 45\%$ respectively. The MB vermicomposted WPS showed the smallest increases of $159 \pm 13\%$ and $24 \pm 2\%$ respectively. Week 2 increases in NH_4^+ were not significantly different between VCUs ($p > 0.05$), however, week 4 values were all highly significantly different ($P < 0.001$). Despite these initial differences, final NH_4^+ contents (mg kg^{-1}) of SB and control WPS after 12 weeks were not significantly different ($P > 0.05$); although, both were significantly below that observed for the MB vermicomposted WPS ($p < 0.001$).

The level of nitrate (NO_3^-) only rose within the SB VCU and control VCU processed WPS. By week 8 the level of NO_3^- within the SB vermicomposted WPS had increased by 2647 mg kg^{-1} , significantly higher than the MB vermicomposted or control WPS ($p < 0.001$, post ANOVA Tukey-Kramer multiple comparisons test), suggesting a more rapid onset of nitrification. During maturation (week 12) the SB vermicomposted WPS increased to a value 6092 mg kg^{-1} higher than for fresh WPS ($p < 0.0001$).

Nitrification was observed within the control WPS at week 12 with an increase of 575 mg kg^{-1} ($p < 0.05$); significantly less than observed for the SB vermicomposted WPS ($p < 0.0001$).

At week 12, due to large levels of nitrification, $\text{NH}_4^+/\text{NO}_3^-$ ratios of SB vermicomposted and control WPS were low, at $0.058 \pm 0.004:1$ and $0.618 \pm 0.348:1$ respectively. These were both significantly below the $\text{NH}_4^+/\text{NO}_3^-$ ratio of $174 \pm 39:1$, recorded for the MB vermicomposted WPS.

5.3.4 Other macro–nutrients (P, K, Mg) including water soluble forms

Table 30 – Phosphorous (P), potassium (K) and magnesium (Mg) contents

VCU	Sample (week)	Total		Water soluble			Nutrient ratios			
		P %	K %	P mg kg ⁻¹	K mg kg ⁻¹	Mg mg kg ⁻¹	C/P :1	C/K :1	N/P :1	N/K :1
Fresh	0	0.54±0.02	0.33±0.01	296±33	1277±40	360±11	83±1	134±4	5.0±0.2	8.1±0.4
SB	2	0.78±0.03	0.46±0.02	727±88	2145±196	397±29	54±2	91±4	4.6±0.3	7.8±0.1
	4	0.95±0.05	0.69±0.08	845±46	2380±51	156±12	42±3	58±7	3.6±0.3	4.9±0.4
	6	0.95±0.05	0.49±0.03	745±14	2646±137	266±11	41±2	79±6	2.7±0.4	5.2±0.2
	8	0.95±0.02	0.42±0.02	639±41	2314±124	477±34	39±2	87±7	2.9±0.2	6.6±0.1
	12	0.90±0.03	0.39±0.01	445±32	2065±71	731±8	38±2	87±3	2.8±0.1	6.3±0.1
MB	2	0.81±0.02	0.36±0.02	579±40	1738±147	167±17	50±2	112±6	3.9±0.2	8.9±0.2
	4	0.88±0.02	0.40±0.04	655±65	2134±108	219±29	44±3	96±10	3.4±0.4	7.5±0.2
	6	0.97±0.05	0.53±0.02	647±21	2342±100	233±13	39±2	71±3	3.1±0.2	5.6±0.1
	8	1.06±0.08	0.44±0.02	918±51	2083±138	206±8	35±3	86±4	3.0±0.3	7.4±0.1
	12	1.12±0.07	0.49±0.05	994±83	2341±219	258±22	33±2	74±9	2.3±0.2	5.2±0.3
control	2	0.89±0.03	0.56±0.02	1080±132	2501±114	357±51	47±2	76±3	4.6±0.3	7.3±0.0
	4	1.05±0.05	0.61±0.01	1652±104	3326±79	191±4	39±2	66±2	3.6±0.3	6.2±0.1
	6	1.17±0.03	0.53±0.04	1351±82	3025±110	197±24	34±1	74±7	3.1±0.2	6.9±0.3
	8	1.08±0.01	0.57±0.02	1131±91	3257±128	260±13	37±1	69±2	3.4±0.1	6.4±0.1
	12	0.96±0.12	0.52±0.06	205±23	3347±207	340±24	39±1	72±10	2.6±0.6	4.8±0.0

Of the initial content of total P in the fresh WPS, only 5.5% was present in water soluble (available) form (0.03%DM); whereas 38.7% of total K was present in water soluble form (0.13%DM).

During the first 4 weeks active vermicomposting there was a significant ($p<0.001$) increase in total P of 0.41%DM ($22\pm0.6\%$ ash) within the SB vermicomposted WPS. However, by week 12 there was a significant net reduction in total P of $17\pm0.4\%$ ash from its initial value ($p<0.01$). Changes in total P coincided with a net increase in water soluble P of $99\pm4.7\%$ ash at week 4, and an overall reduction of $25\pm1.3\%$ ash by week 12. This also coincided with significant ($p<0.001$) increases in total K ($45\pm1.8\%$ ash) and water soluble K ($29\pm0.6\%$ ash) within the SB VCU processed WPS by week 4; and an overall reduction of $42\pm0.8\%$ ash total K ($p<0.001$) and $20\pm0.4\%$ ash reduction in water soluble K by week 12.

Total P within the MB vermicomposted WPS at week 2 (casts) was not significantly different from initial levels ($p>0.05$), corrected for dry matter losses (%ash). Total P increased gradually during vermicomposting, reaching a final value $17\pm0.5\%$ ash above initial levels ($p<0.01$). This coincided with a net increase in water soluble P of $37\pm1.9\%$ ash at week 2, which continued to rise in the absence of earthworms to $89\pm4.7\%$ by week 12. Concurrent with total and water

soluble P increases was a total K loss of $25 \pm 0.7\%$ after 2 weeks ($p < 0.01$) and $17 \pm 0.6\%$ by week 12; with no significant changes in water soluble K ($p > 0.05$).

Within the control VCUs total P had increased by $37 \pm 0.9\%$ by week 2 ($p < 0.01$), and ranged between 36-45% above initial levels until week 12. After 4 weeks maturation (week 8 to 12) total P fell from 36% above initial level, back to the original level (corrected for DM loss). Changes in total P within the control WPS coincided closely with increases in water soluble P, which ranged between 160 to 306% above initial levels for weeks 2 to 8. This fell to 59% below initial levels by week 12. Total K also increased ($39 \pm 0.8\%$) within the control WPS over the first 2 weeks ($p < 0.01$). After 12 weeks this also had fallen to a level that was not significantly different from fresh WPS. Water soluble K increased within the control WPS ($90 \pm 1.7\%$ week 4), and remained significantly ($p < 0.01$) above original levels ($55 \pm 1.6\%$ by week 12).

Reductions in water soluble Mg occurred in all VCU processed WPS over 8 weeks active vermicomposting. The MB processed WPS showed the highest reduction at 66%, followed by the control at 51%, and the SB processed WPS at 25%. However, within the SB vermicomposted WPS, water soluble Mg level rose again to give no overall net reduction (%).

Final concentrations of total P and K (%DM) were not significantly different between treatments. However, differences in final concentrations of water soluble P were significant between all treatments ($P < 0.05$, post ANOVA Tukey-Kramer multiple comparisons test); MB processed WPS possessed an approximately 2-fold greater concentration than the SB processed WPS, which in turn contained an approximately 2-fold greater concentration than the control. Water soluble K concentrations were not significantly different between the MB or SB processed WPS ($p > 0.05$); which were both significantly below the control ($p < 0.01$).

Final water soluble Mg concentration was highest in the SB vermicomposted WPS which was significantly above ($P < 0.0001$) both the MB and control WPS, which did not differ significantly ($p > 0.05$).

All nutrient ratios fell significantly below ($p < 0.001$) initial values for all VCUs after 8 weeks of vermicomposting; showing that relative losses of carbon and TKN exceeded reductions of P and K within WPS processed by all VCUs. No significant differences in final nutrient ratios were observed between treatments, except for N/K ratios, which was significantly higher in the SB vermicomposted WPS than in either the MB or control WPS ($p < 0.01$).

5.3.5 Changes in acidity/alkalinity (pH) and electrical conductivity (EC).

Table 31. – Changes in pH during active vermicomposting

VCU	Sample (week)	pH	H ⁺	EC μS cm ⁻¹
Fresh	0	7.2±0.0	7.3 × 10 ⁻⁸	1350±66
SB	2	6.0±0.0	1.1 × 10 ⁻⁶	1300±87
	4	7.3±0.1	5.3 × 10 ⁻⁸	1252±104
	6	6.8±0.0	1.7 × 10 ⁻⁷	650±23
	8	6.5±0.2	3.2 × 10 ⁻⁷	970±68
	12	6.2±0.0	6.7 × 10 ⁻⁷	1913±145
MB	2	7.1±0.1	8.7 × 10 ⁻⁸	830±31
	4	6.8±0.1	1.5 × 10 ⁻⁷	670±21
	6	7.3±0.0	4.7 × 10 ⁻⁸	613±10
	8	6.8±0.0	1.7 × 10 ⁻⁷	548±9
	12	7.2±0.0	6.3 × 10 ⁻⁸	600±9
Control	2	6.1±0.2	7.9 × 10 ⁻⁷	1380±79
	4	7.5±0.0	3.5 × 10 ⁻⁸	1988±83
	6	6.8±0.1	1.5 × 10 ⁻⁷	983±19
	8	6.8±0.0	1.8 × 10 ⁻⁷	658±11
	12	7.9±0.1	1.2 × 10 ⁻⁸	825±32

A significant decrease in pH relative to initial values occurred within the SB processed and control WPS after 2 weeks ($p<0.001$), equating to approximately 16-fold and 11-fold increases in $[H^+]$, respectively. These pH values were significantly below that observed for the MB vermicomposted WPS (casts) ($p<0.001$), which were not significantly different from their initial value.

However, at week 4 pH within the SB processed and control WPS increased significantly to 7.3 and 7.5 respectively ($P<0.001$). This was concurrent with large increases in NH_4^+ ions (section 5.3.3).

Final pH value (week 12) within the SB vermicomposted WPS was significantly below that for MB and control WPS ($P<0.001$). The final pH within the control WPS (pH 7.9) was significantly higher than at any other stage during the 12 week experimental period ($p<0.001$). In the MB vermicomposted WPS final pH (Ph 7.2) was not significantly different from the initial value ($p>0.05$).

EC within the SB VCUs did not change significantly until week 6 of active vermicomposting, where it showed a mean reduction of $700\text{ }\mu\text{S cm}^{-1}$ (52%) ($p<0.001$). This coincides with a large reduction in NH_4^+ (section 5.3.3). By week 12 EC rose again to a level significantly above that for fresh WPS ($p<0.01$). This was likely to be due to a concurrent increase in NO_3^- ions (section 5.3.3).

Pearson's correlation analysis showed a significant correlation between EC and $\text{NH}_4^+ + \text{NO}_3^-$ ions for all samples taken from all treatments during the 12 week experimental period (Pearson $r = 0.91$; $p < 0.0001$; $R^2 = 82.6\%$; 60 x,y-pairs). Pearson's correlation analysis of EC against the total concentration of all water soluble nutrients ($\text{NH}_4^+ + \text{NO}_3^- + \text{P}^{3-} + \text{K}^+ + \text{Mg}^{2+}$), recorded for all samples from all experimental treatments, showed a slightly weaker correlation (Pearson $r = 0.88$; $p < 0.0001$; $R^2 = 77.0\%$).

After 2 weeks active vermicomposting the MB vermicomposted WPS (casts) had a significantly lower EC than either the SB vermicomposted or control WPS ($p < 0.001$); and were significantly below the EC of fresh WPS ($p < 0.001$). EC within the WB vermicomposting remained significantly below the SB vermicomposted and control WPS after 12 weeks, and below fresh WPS ($p < 0.01$). This reflects an overall net decrease in water soluble ions during vermicomposting, especially NH_4^+ and NO_3^- (section 5.3.3.), this being supported by Pearson's correlation analysis.

5.3.6 Earthworm growth and reproduction

Table 32 - Earthworm growth and reproduction after 8 weeks vermicomposting

Growth					
Initial total worm biomass per VCU (g)	Initial individual worm biomass (g)	Final total worm biomass per VCU (g)	Final individual worm biomass (g)	Total Mortality (%)	
500±5	1.4±0.1	683±18	2.5±0.1	22.3±1.7	
Reproduction					
Total number of Cocoons per VCU	Individual cocoon wgt. (mg)	Total number of hatchlings per VCU	Individual hatchling wgt. (mg)	Cocoon viability (%)	Hatchlings per cocoon
123±10	38±1	38±5	27±2	83.6±2.3	1.05±0.03

An increase in total worm biomass was observed in all single-batch VCUs, with a mean increase of 183g wet mass per VCU. This equates to a mean increase of 36.6% over the initial inoculation mass.

This biomass increase was due to an increase in average individual worm biomass from 1.4±0.1 to 2.5±0.1 g (78.6%). This is equivalent to an overall average growth rate of 137.5 mg g worm⁻¹ wk⁻¹ (19.6 mg worm⁻¹ d⁻¹).

A mortality rate of 22.3% was calculated from a mean reduction in earthworm number from 357±33 (initial inoculation) to 273±11 per VCU (8 weeks vermicomposting); this is equivalent to 1.5 worm deaths per VCU per day.

Very little reproduction had occurred during the 8 weeks, with on average 123 cocoons and 38 hatchlings produced per VCU; equivalent to <0.05 cocoons worm⁻¹ week⁻¹, and <0.02 hatchlings worm⁻¹ week⁻¹.

Of the cocoons that were produced, 83.6% hatched over the 6 week incubation period at 21±1°C; with an average of 1.05 hatchlings cocoon⁻¹.

5.4 Discussion

The initial moisture content of the WPS (77.8%) rose sharply over the first 2 weeks vermicomposting to levels of between 82-84%. The moisture level remained >80% for the rest of the 8 week active vermicomposting period. Such moisture levels can be inhibitory to the decomposition process, and are in excess of suggested limits, (e.g. 75%, Miller 1991). However, such moisture levels are near optimal for the growth and reproduction of *D. veneta* earthworms (e.g. 75-80%, Viljoen *et al.* 1991 and 67-84%, Muyima *et al.* 1994). The more rapid drying of casts compared with the control WPS, during maturation, was also experienced by Hartenstein & Hartenstein (1981). The MB casts did not dry as rapidly due to a loss of cast structure during maturation.

Temperature within the VCU systems was maintained at $20.8 \pm 0.7^\circ\text{C}$, which is optimal for the growth and reproduction of *D. veneta* (e.g. 20-25°C, Fayolle *et al.* 1997). However, this is below temperatures considered optimal for organic decomposition during composting (e.g. 45-60°C, Bardos & Lopez-Real 1991).

An overall earthworm growth rate of $19.6 \text{ mg worm}^{-1} \text{ d}^{-1}$ was observed over 8 weeks within the SB VCUs. This is comparable to $27.2 \text{ mg worm}^{-1} \text{ d}^{-1}$ observed for *D. veneta* cultured using an excess supply of paper sludge, over 12 weeks at 25°C (Fayolle *et al.* 1997); and $11.9 \text{ mg worm}^{-1} \text{ d}^{-1}$ recorded for *D. veneta* on cattle manure over 200 days (Viljoen *et al.* 1992).

A mortality rate of 22.3% was recorded within the SB VCUs over 8 weeks. This compares favourably to the high mortality (<50 to >75%) observed by Elvira *et al.* (1997) when utilising *Eisenia andrei* to process paper sludge mixed with sewage sludge, pig slurry and poultry slurry, although Fayolle *et al.* (1997) observed a much lower mortality ($\leq 6\%$) of *D. veneta*, utilising paper sludge as the sole food source.

The SB earthworm reproduction rate was very low, as expected with the high stocking density used, with an overall reproduction rate of $<0.05 \text{ cocoons worm}^{-1} \text{ week}^{-1}$ (c.f. $2.0 \text{ cocoons worm}^{-1} \text{ week}^{-1}$, Lofs-Holmin 1985; 1.6, Edwards 1988; 1.96 at 25°C and 1.19 at 20°C, Viljoen *et al.* 1992; and 4.5 at 20°C on paper sludge, Fayolle *et al.* 1997). Cocoons produced within the SB VCUs had a mean individual mass of 38 mg, comparable with other studies (27.2-29.6 mg, Fayolle *et al.* 1997). Cocoon viability (83.6%) compared favourably to 37.8% observed by Viljoen *et al.* (1992), and was similar to a minimum of 84% observed by Fayolle *et al.* (1997). Polyembryony was typically low, $1.05 \text{ worms cocoon}^{-1}$, as seen in other investigations ($1.1 \text{ worm cocoon}^{-1}$, Edwards 1988; 1.1 at 25°C and 1.2 at 15°C, Viljoen *et al.* 1992; 1.02, Fayolle *et al.* 1997).

It appears from the earthworm growth and reproduction findings that the stocking density used was approaching the carrying capacity of the system, but not high enough to cause a population crash. Although worm numbers decreased, individual worm biomass increased, providing an adequate total earthworm biomass to process all WPS feedstock into casts within 8 weeks active vermicomposting.

The conversion of all WPS into casts within the SB vermicomposting VCU equates to a processing (ingestion) rate of approximately $0.17 \text{ g WPS worm}^{-1} \text{ day}^{-1}$, allowing for the removal of WPS during sampling. A mean of 56% of each 2 kg batch of WPS was processed within the MB vermicomposting systems over each 2 week period (estimated from the mass of unprocessed WPS remaining); This equates to a processing rate of $0.17 \text{ g WPS worm}^{-1} \text{ day}^{-1}$, identical to that of the SB system. These processing rates are low compared to those observed in some studies: $0.5\text{-}1.0 \text{ g worm}^{-1} \text{ day}^{-1}$ (*Lumbricus rubellus*) (Pierce 1978); $0.3\text{-}0.5 \text{ g worm}^{-1} \text{ day}^{-1}$ (*D. veneta*) (Lofs-Holmin 1985); and $0.25 \text{ g worm}^{-1} \text{ day}^{-1}$ (Hartenstein *et al.* 1981). However, these processing rates compare favourably to other trials: $0.13 \text{ g worm}^{-1} \text{ day}^{-1}$ (*E. fetida*) using a higher bed stocking density of $8.8 \text{ kg worms m}^{-2}$ when vermicomposting solid municipal waste (Pincince *et al.* 1981).

Significant levels of TOM decomposition were achieved in all experimental VCUs ($p < 0.01$).

After 2 weeks vermicomposting the MB WPS showed a very significant decrease in TOM ($p < 0.001$); the processed WPS analysed at this point consisted solely of earthworm casts. A continued significant ($p < 0.001$) decrease in TOM to week 4 within the MB produced casts clearly showed that they were not stable, in terms of TOM. This was supported by a significant ($p < 0.05$) reduction in TOM during the 4 week maturation of SB vermicomposted WPS between weeks 8 and 12; the SB vermicomposted WPS consisting nearly entirely of casts.

The activation of dormant micro-organisms in soil during conversion into earthworm casts (Lavelle & Gilot 1994), as well as increases in microbial respiration within casts (Scheu 1987), may suggest reasons for the continued TOM decomposition within the WPS casts. Hartenstein & Hartenstein (1981) found *Eisenia foetida* casts of activated sludge showed a proliferation of microbial biomass, although respirometry analysis (O_2 consumption rate) showed the casts were stable 2 weeks after production.

Non-linear regression analysis showed the most rapid rate of TOM loss occurred within the MB produced casts, indicated by a significantly greater decay rate constant and half-life ($p < 0.01$). General observation revealed that, on incubation in the absence of earthworms, MB WPS casts lost their physical structure, becoming increasingly water-logged and compacted by

week 12. It was likely that anaerobic conditions developed within the casts during incubation, reducing TOM decomposition.

During vermicomposting for two weeks, the MB vermicomposted WPS completed 50% of its total TOM losses (7%DM) in 1.33 weeks, a loss of 5.3%DM per week. This was a 2-fold greater rate of decay ($p < 0.01$) than observed for the control during the completion of 50% of its total TOM losses (6%DM in 2.33 weeks), a decay rate of 2.6%DM per week.

These findings are broadly comparable to a 2-fold increase in the rate of stabilisation (reduction in TOM) in sewage sludge observed over 20 days vermicomposting by Neuhauser *et al.* (1988); and Hartenstein & Hartenstein (1981) who, when vermicomposting activated waste water sludge, observed a 1.3-fold increase in 'mineralisation' rate, with a decay rate of 4.0%DM per week during the first 2 weeks, compared to 2.3%DM without worms. Comparable rates of decomposition were achieved during the early stages of MB vermicomposting of WPS (4.2%DM per week over the first 2 weeks).

SB vermicomposting showed a lower decomposition rate (2.5%DM per week) during the first 2 weeks, similar to that of the control VCUs (2.1%DM per week). SB vermicomposted WPS after 2 weeks, consisting of a mixture of few casts and mainly unprocessed WPS, showing no significantly different TOM loss from the control ($p > 0.05$). TOM losses within the SB processed WPS were not significantly different from the control until 4 weeks vermicomposting ($p < 0.05$).

Non-linear regression analysis showed a more rapid cessation of TOM loss occurred within the control WPS than the SB vermicomposted WPS, indicated by a significantly greater decay rate constant and half-life ($p < 0.05$). General observation revealed that during the decomposition of WPS inside the control VCUs it lost physical structure, becoming increasingly water-logged and compacted by week 12. It was likely that anaerobic conditions developed within the WPS, reducing TOM decomposition. Vermicomposted WPS within the SB VCUs remained less compact and more porous through continuous agitation by the earthworms. A more prolonged TOM loss within the SB vermicomposted WPS (*i.e.* during maturation) occurred, producing a significantly lower final TOM content (61.6%DM) than the MB treated (66.1%DM) or control (67.8%DM) WPS ($p < 0.01$). This may have been due the increased aeration within the less compact SB-vermicomposted WPS, indicated by its more rapid desiccation (see above).

Both SB and MB vermicomposted WPS showed a significantly lower final TOM content than the control after 12 weeks ($p < 0.05$) suggesting increased decomposition effected by the presence of earthworms. Increased TOM losses due to the activity of earthworms have also

been observed for green waste (Frederickson *et al.* 1997); sheep manure and cotton wastes (Albenell *et al.* 1988); and cattle manure (Mitchell 1997).

The final mean TOM content for the SB vermicomposted WPS (61.6%DM) appears higher than usually obtained for traditionally composted materials: 36.9%DM - cattle slurry co-composted with rice hulls for 36 weeks, 8 weeks active composting (cylindrical adiabatic reactor) plus 28 weeks curing (Genevini *et al.* 1996); 32.6% - vermicomposting green waste for 8 weeks (Frederickson *et al.* 1997); and 39.5%DM - a mixture of waste paper sludge, chicken litter and yard waste (8:2:1 by volume) after 27 weeks composting, 7 weeks active composting (aerated static pile) plus 20 weeks maturation (Sesay *et al.* 1997). Worm-worked animal manures can contain final TOM levels in the range of 32 to 52% (Lofs-Holmin 1985).

However, total TOM losses of 62.3%ash obtained for the SB vermicomposted WPS, 54.2%ash MB vermicomposted, and 50.6%ash for the control, are broadly comparable to losses observed for other materials (composted using varying traditional composting methods over varying duration): 60% - food waste and leaves (Michel *et al.* 1996); 52% cattle manure (Tarre *et al.* 1987); 75% - green waste (Frederickson *et al.* 1997); and 55% - a mixture of waste paper sludge, chicken litter and yard waste (Sesay *et al.* 1997).

The initial C/N ratio of fresh WPS (16.5:1) was within a suggested range of 15:1-35:1 for optimal earthworm nutrition during vermicomposting (Neuhauser *et al.* 1980b). This broadly falls within proposed ranges for optimal decomposition, but with a lower minimum value; 30:1-35:1 (Gray *et al.* 1971); 26:1-35:1 (Poincelot 1975); 30:1-50:1 (Taiganides 1977); 20:1-30:1 (Jansson & Persson 1982); 20:1-35:1 (Haug 1993).

A mean overall loss in total nitrogen of 54.7%ash within the SB vermicomposted WPS over 12 weeks indicated an excess of nitrogen content. Overall losses of nitrogen in the MB and control WPS were 46.6%ash and 46.2%ash respectively. This suggests the prolonged presence of earthworms in the SB VCU compared to the MB VCU significantly increased nitrogen loss ($p < 0.01$). Increased nitrogen losses due to the presence of earthworms were also observed by Hartenstein & Hartenstein (1981) and Haimi & Huhta (1987).

Nitrogen loss may have occurred via volatilisation, as although temperature and pH were low, high levels of ammonium were observed within the WPS of all VCUs at weeks 2 and 4 of the experiment. The presence of earthworms seemed to have a significant negative effect on the amount of ammonium present. The MB and SB WPS contained significantly lower amounts of ammonium than the control at week 4 ($p < 0.001$). High ammonium levels persisted in the MB WPS, suggesting anaerobic conditions formed within the WPS in the absence of earthworms. High moisture contents of the WPS during the experimental period also made the loss of nitrogen, through leaching, possible.

Macro-nutrient (N, P, K) reductions within the vermicomposted WPS would also have been due to their uptake by earthworms during growth. Mitchell (1997) observed nutrient extractions from cattle manure by earthworms of 18% N and 2% P during vermicomposting; and Hartenstein & Hartenstein (1981) also observed the uptake of minerals (N,P,K) by earthworms during activated waste water sludge vermicomposting. This may also partly explain the highly significant ($p < 0.001$) increases in N, P and K losses within the MB vermicomposted WPS at week 2, and within the SB vermicomposted WPS at week 8 ($p < 0.001$). It is also important to note that the earthworm mortality recorded (22%) would have reintroduced some of these nutrients during active vermicomposting.

Despite losses in nitrogen during vermicomposting, the C/N ratio of the WPS significantly reduced within SB and WB vermicomposting VCU's to 13.7:1 and 14.1:1 respectively. These figures fall within suggested levels for stabilised materials: 5:1-20:1 (Hitai *et al.* 1983), $\leq 30:1$ (ORCA 1992).

N/P ratios of between 5:1 and 20:1 for microbial cells suggest a composting feedstock should contain phosphorous at levels between 5 and 20% of the concentration of nitrogen (Alexander 1977). Gray *et al.* (1971) suggest a C/P ratio of 75:1 to 150:1.

The N/P and C/P ratios of the fresh WPS were 5:1 and 83:1, suggesting P content was not a limiting factor.

Large levels of water extractable K within the WPS of all treatments after 12 weeks suggest that K was present in excess of microbial requirements, *i.e.* immobilisation was less than mineralisation (Swift *et al.* 1979).

The production of large levels of nitrate within the SB composted WPS at weeks 8 (2656 mg kg⁻¹DM) and 12 (6111 mg kg⁻¹DM) is often used as an indicator of favourable composting conditions (Miller 1992) and the onset of maturation (Finstien & Miller 1985) within traditional composting systems. Increased nitrification effected by earthworms has also been observed during the vermicomposting of other wastes, such as, cattle manure, pig manure, and potato waste (Edwards & Burrows 1988); and sewage sludge mixed with pine bark (Haimi & Huhta 1987).

The acidic first stage of the nitrification process (The Open University 1996a) may explain significant ($p < 0.05$) decreases in pH observed for the SB vermicomposted WPS during weeks 8 and 12. Hartenstein & Hartenstein (1981) also observed reductions in pH during vermicomposting (activated sludge), and suggested the production of CO₂, organic acids, and to a lesser extent, humic acids were potential causes. Haimi & Huhta (1987) observed lower pH levels in worm-worked sewage sludge than unworked sludge.

A final level of NH_4^+ (0.036%DM) within the SB vermicomposted WPS after 12 weeks also indicated a stabilised material, falling below a suggested limit of 0.04%DM (Forster *et al.* 1993, and Bernal *et al.* 1998). A $\text{NH}_4^+/\text{NO}_3^-$ ratio of 0.058:1 also suggested the maturity and stability of the final SB vermicomposted WPS, falling well below the suggested limit of 0.16:1 (Bernal *et al.* 1998).

No nitrification was observed within the MB vermicomposted WPS, and a high level of NH_4^+ (1115 mg kg⁻¹DM) was retained; this in turn results in a high final $\text{NH}_4^+/\text{NO}_3^-$ ratio (174:1) at week 12. Nitrification, an aerobic process, may have been inhibited by anaerobic conditions within the MB produced WPS casts (created by a lack of physical structure and high moisture contents, see previous discussion). This may also explain the large build up of NH_4^+ , which requires sufficient aeration (ventilation) to facilitate volatilisation as NH_3 .

After 8 weeks, TOM loss rates (mean of 1.79%DM wk⁻¹) did not indicate that the WPS in vermicomposted SB VCU had reached stability. A significant ($p < 0.001$) loss in TOM (9.4%ash) (mean of 1.28%DM wk⁻¹) was observed during maturation (weeks 8 to 12), this being 71% of the level seen in the first 8 weeks, during active vermicomposting.

5.5 Conclusions

The WPS provided was a substrate suitable (in nutritional terms) for vermicomposting with *D. veneta*. *D. veneta* increased in biomass, despite being cultured at high population densities, but net reproduction rates were consequently very low. The high nutritive value of WPS is emphasised by the observation of this increased biomass despite relatively low ingestion rates.

The decomposition and stabilisation rates of commingled WPS were significantly improved by vermicomposting with *D. veneta*, compared to incubating WPS without *D. veneta* under similar environmental conditions. Reduction in TOM was initially accelerated by the higher ratio of earthworms to WPS in the MB system compared to SB system.. However, the stabilisation of casts after separation from earthworms appeared to be restricted by a loss of physical structure and moisture increases, resulting in anaerobic conditions within the vermicompost.

WPS earthworm casts were not fully stabilised immediately after production for both systems.

Although nitrogen loss was also increased by the presence of earthworms, nitrification was also increased, with significantly higher levels of nitrate within the final vermicomposted WPS. Low ammonium levels and high nitrate levels suggested a stabilised product had been produced after 8 weeks, despite a high final TOM content and continued TOM losses during maturation.

It appears that the final vermicomposted WPS could have considerable value as a fertiliser, and therefore could provide a useful component of plant growth media (investigated in Chapter 6).

6. Windrow-composted and vermicomposted waste paper sludge as components of plant growth media.

6.1 Introduction

The 'quality' of the final composted products is crucial to any composting operation in terms of their economic and environmental value. In vermicomposting, some studies have suggested the composted products would provide more economic benefit than the resulting worm biomass which was also produced (Fieldson 1985). Beneficial economic and environmental effects of adding composted organic matter to soil have also been outlined (Steffen 1979). Knight (1991) has discussed the environmental importance of finding alternatives for peat, which have included vermicompost.

Although the quality of composts can be assessed by analysing their physicochemical characteristics (*e.g.*, Levi-Minzi *et al.* 1992; Lamim *et al.* 1996 & 1998), the value of biological evaluation of compost maturity has long been established (Zucconi *et al.* 1981).

The effect of compost stability on plant growth has been investigated in several studies (Hadar *et al.* 1985; Hartz & Giannini 1998; Inbar *et al.* 1993; Keeling *et al.* 1994 & 1995; Wong & Chu 1985).

Numerous studies have also been conducted into the value of fresh and composted paper sludges on plant growth as both soil amendments (Feagley *et al.* 1994; Kraske & Fernandez 1993; Phillips *et al.* 1997; Voundi-Nkana *et al.* 1999), and as components of plant growth media (Chong & Cline 1993 & 1994; Chong & Hanersma 1996; Bellamy *et al.* 1995; Tripepi *et al.* 1996).

However, few comparisons between vermicomposted and conventionally composted products have been conducted (Haimi & Huhta 1987; Subler *et al.* 1998).

Haimi & Huhta (1987) investigated conventionally composted and vermicomposted products derived from identical wastes. However, due to the small scale involved, it is likely that composting without earthworms was carried out under sub-optimal conditions, *i.e.* poor aeration and heat retention. The composted products were evaluated on the basis of their physicochemical characteristics only, and significant differences between composts were minimal.

Subler *et al.* (1998) also observed overlap in the physicochemical composition of composted and vermicomposted materials, and evaluated products using plant growth experiments.

However, they investigated composts and vermicomposts derived from various waste types obtained from different locations, making direct comparisons unreliable.

During this study, compost and vermicompost derived from an identical waste paper sludge (obtained from previous experiments [Chapters 4 & 5]) was used for investigation into the differences between composted products produced by the two differing composting processes.

The composts physical and chemical properties were analysed, and the germination and growth of radish was used for the biological evaluation of the composts as components of plant growth media.

6.2 *Materials and methods*

6.2.1 Sampling and physicochemical analyses of plant growth media

Following composting and vermicomposting of the commingled waste paper sludge, one composite sample was taken from each experimental plant growth medium. Approximately 10 one litre samples of each medium were amalgamated and thoroughly mixed, from this a one-litre sub-sample was taken (Leege & Thomson 1998).

The following determinations were conducted for all experimental growing media samples:

- 1) Bulk density (BD) - g l^{-1} , wet mass (WM) and dry mass (DM);
- 2) Air-filled-porosity (AFP) - %WM;
- 3) Electrical conductivity (EC) - $\mu\text{S cm}^{-1}$ at 25°C ;
- 4) Acidity/alkalinity (pH) - standard units ($-\log[\text{H}^{+}]$);
- 5) Total macronutrients: nitrogen (N), phosphorus (P), and potassium (K) - %DM;
- 6) Water-extractable nutrients: ammonium $[\text{NH}_4^{+}]$, nitrate $[\text{NO}_3^{-}]$, phosphorus $[\text{H}_2\text{PO}_4^{-}]$, potassium $[\text{K}^{+}]$, sulphur $[\text{SO}_4^{2-}]$, calcium $[\text{Ca}^{2+}]$, magnesium $[\text{Mg}^{2+}]$, iron $[\text{Fe}^{2+}]$, sodium $[\text{Na}^{+}]$, and chloride $[\text{Cl}^{-}]$ - mg l^{-1} WM and mg kg^{-1} DM.
- 7) Total potentially toxic elements (PTE): aluminium (Al), arsenic (As), copper (Cu), zinc (Zn), lead (Pb), nickel (Ni), chromium (Cr), cadmium (Cd), mercury (Hg) - ppm DM and mg l^{-1} DM (conducted for all individual components of experimental plant growth media).

Levington Agriculture laboratories (Ipswich, Suffolk, UK) conducted all determinations.

6.2.2 Plant cultivation trials

Experimental plant growth media

Windrow-composted waste-paper sludge (W-WPS) and vermicomposted waste-paper sludge (VCU-WPS), obtained from previous experiments (see Chapters 4 & 5), were investigated both with and without coir-amendment.

Comparative media included:

- 1) Immature green-waste compost (GW), composted for 8 weeks at local open-air, mechanically-turned, windrow-composting facility (Shanks & McEwan, Newton-Longville), with and without coir-amendment;
- 2) a commercial peat-based multipurpose potting medium, Levington's multipurpose-compost (LM), a market leading peat-based potting medium;
- 3) 100% coir controls (C). Coir was virgin fibre without fertiliser amendment.

W-WPS, VCU-WPS and GW were all sieved to a particle size of ≤ 10 mm, as recommended during the use of peat in horticulture (BSI 1990) and planting composts (DETR 1999b).

Coir (dust and fibre obtained from the fruit mesocarp of the palm - *Cocos nucifera* L.) was used in amended media, providing an alternative to peat (Knight 1991). Amended media derived from W-WPS, VCU-WPS, and GW composts (coded as W2, V2, and G2, respectively) were blended with coir in a ratio of 1:1 (compost:coir) by volume.

Test species

The test species employed was radish (*Raphanus sativus* L.), cv. Sparkler 3. Radish is considered an excellent experimental plant, commonly used to assess plant responses to various environmental stresses (Kostka-Rick & Manning 1993). Radish was a test plant recommended in the Organization for Economic Cooperation and Development guidelines for the testing of chemicals (OECD 1984).

Fertilization and watering regime

During each watering period half of the replicates of each experimental plant growth media ($n = 8$) were supplied with a nutrient solution and half ($n = 8$) with distilled water, at a rate of 0.1 l d^{-1} . The fertiliser solution consisted of a commercial plant feed (Chempak, 'Formula 3'; 20:20:20 - N:P:K) diluted to a concentration recommended for frequent application ('every watering'). The dry feed was diluted with distilled water to a concentration of 0.4 g l^{-1} , providing 0.08 g nitrogen (N); 0.07 g phosphorus (P); and 0.07 g potassium (K). Nitrogen was provided in nitric [N_2O_5], ammonical [NH_4^+], and ureic [$(\text{NH}_2)_2\text{CO}$] forms in a ratio of 1.4:1:3.3; phosphorus was provide in a pentoxide [P_2O_5] form; and potassium was provided in oxide [K_2O] form.

The fertiliser solution also provided lower levels of boron (B) - 0.02%, copper (Cu) - 0.01%, iron (Fe) - 0.2%, manganese (Mn) - 0.02%, molybdenum (Mo) - 0.002%, zinc (Zn) - 0.05%, and magnesium (Mg) - quantity not specified.

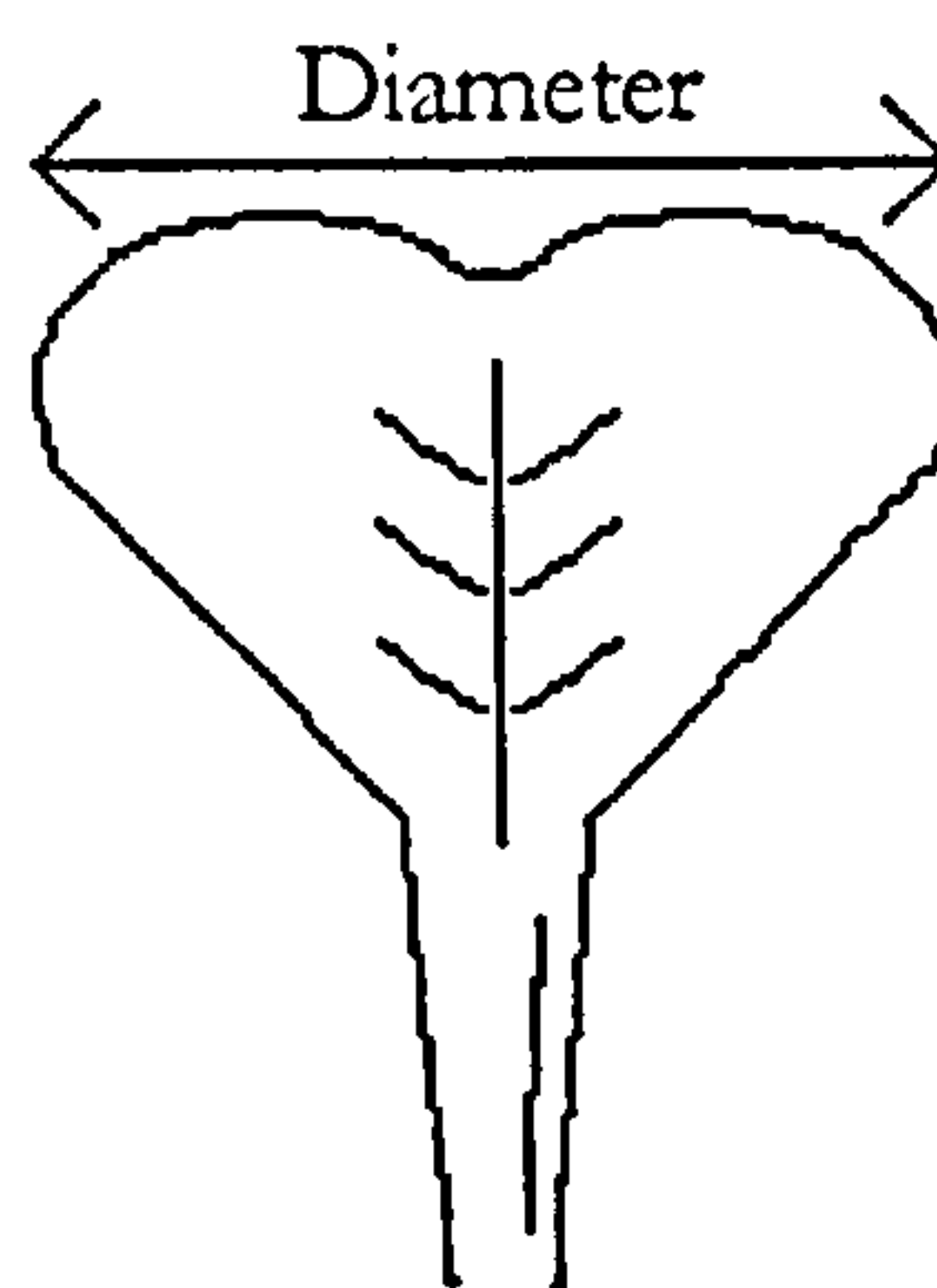
Supplemental macronutrients (N, P and K) were supplied in equal quantities in an attempt to minimise the alteration of nutrient ratios within the plant growth media, allowing for a clearer interpretation of plant growth responses in relation to physicochemical compositions of plant growth media.

Germination and 'early' plant growth

Radish seeds were sieved to 2-3 mm, to reduce potential experimental variations (Kubka *et al.* 1974). For each growing medium, eight randomly selected seeds were planted at a depth of 10 mm, at regular intervals (2.0-2.5cm) within each 0.5 litre (20cm × 5cm × 5cm - length × width × depth) plastic tray, filled with the growing medium under test ($n = 16$). Trays were arranged using a completely-randomised-block design (Clarke 1977). The treatments tested are detailed in Table 1.

The number of emerged seedlings (cotyledons 1cm above the growth media surface) was recorded after 7 (early germination rate) and 14 days (taken as total germination); the diameter of the cotyledons was measured after 14 days as a measure of early plant-growth responses (see figure 33).

Figure 33 - Radish cotyledon



Total plant biomass production and shoot : hypocotyl (S:H) ratio

After 14 days seedlings were thinned to 3 equally-spaced plants per replicate tray, and cultivated for a further 7 days. Plants were harvested, divided into shoot (leaves) and hypocotyl (bulb), dried in a desiccating oven at 60°C for 48 hours (Anon. 1986), and weighed. Environmental stresses tend to cause the continued production of shoot biomass at the expense of the hypocotyl (Cooley & Manning 1987; Kostka-Rick & Manning 1993); therefore, high shoot : hypocotyl (S:H) ratio was considered an indicator of plant-stress and nutrient imbalance. Research shows that root : shoot ratios have been used as indicators of nutrient stress in other plant species (Gerloff & Gabelman 1983).

6.3 Results

6.3.1 Germination and early plant growth

Table 33 - Radish germination and cotyledon diameter ($\bar{x} \pm se$).

Plant growth media	Fertiliser addition (F)	Germination ¹ (%)		Cotyledon diameter (mm)
		7 days (early)	14 days (total)	14 days
VCU-WPS	-	25.0±8.2	85.9±4.4	13.0±0.5
Labelled (V)	F	15.6±5.7	82.8±7.5	12.2±0.4
VCU-WPS (+50% C)	-	79.7±7.1	93.8±3.3	15.7±0.4
Labelled (V2)	F	70.3±8.5	92.2±3.3	14.5±0.4
W-WPS	-	81.3±8.2	93.8±2.4	16.7±0.6
Labelled (W)	F	62.5±4.1	95.3±2.3	15.9±0.5
W-WPS (+50% C)	-	70.3±9.8	93.8±3.3	18.1±0.8
Labelled (W2)	F	78.1±5.2	90.6±3.9	17.7±0.5
Green Waste (GW)	-	1.6±1.6	68.8±9.5	9.3±0.4
Labelled (G)	F	1.6±1.6	59.4±7.4	8.6±0.4
GW (+50% C)	-	46.9±8.8	98.4±1.6	15.3±0.6
Labelled (G2)	F	62.5±8.8	96.9±2.1	13.2±0.3
Lev'ton M. P. (LM)	-	92.2±4.1	92.2±4.1	16.7±0.5
Labelled (L)	F	82.8±5.8	98.4±1.6	15.7±0.5
Coir (C)	-	82.8±6.2	98.4±1.6	15.2±0.4
Labelled (C)	F	81.3±4.1	93.8±2.4	16.4±0.5

Figure 34 - Germination (7 days)

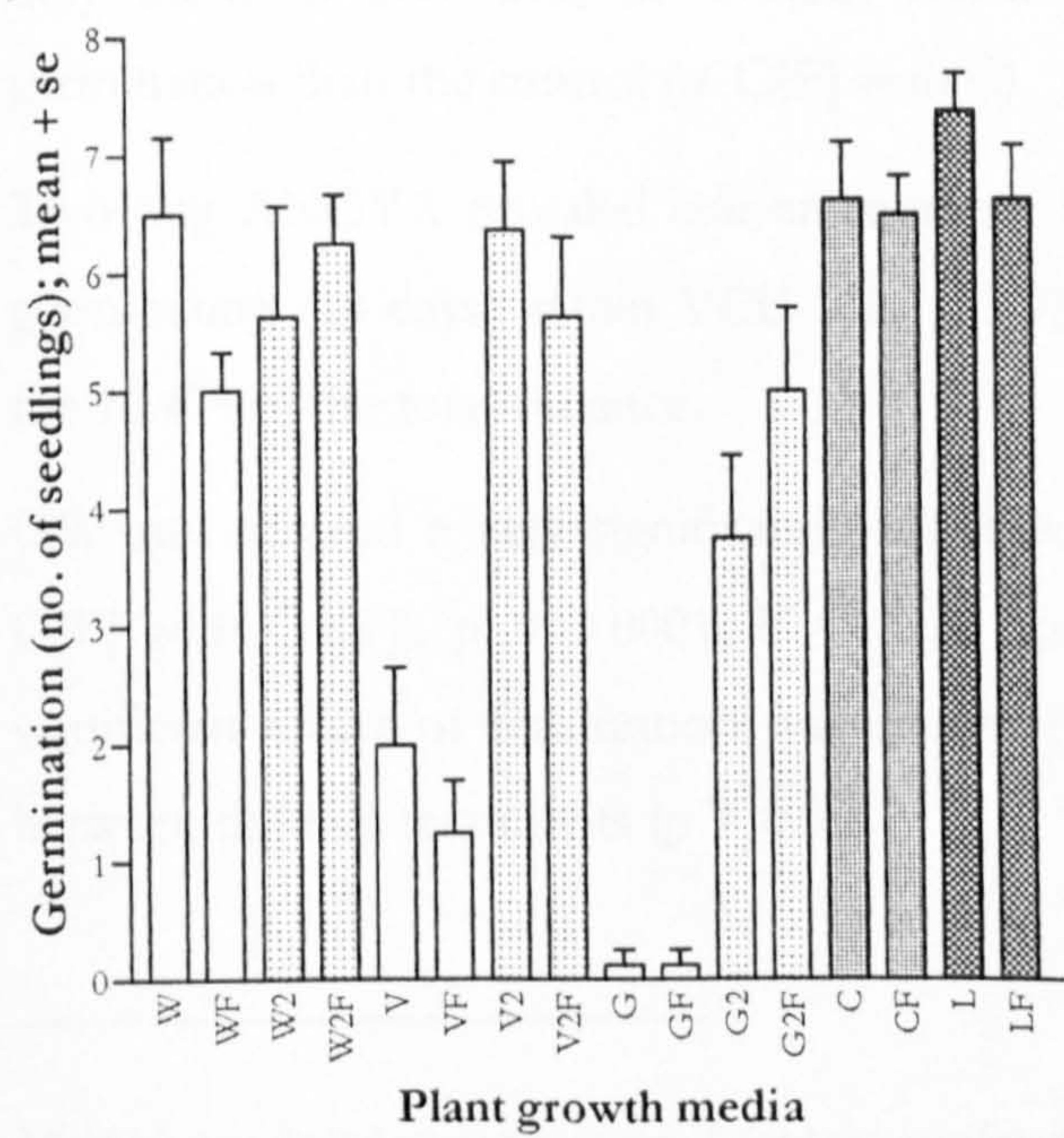


Figure 35 – Germination (14 days)

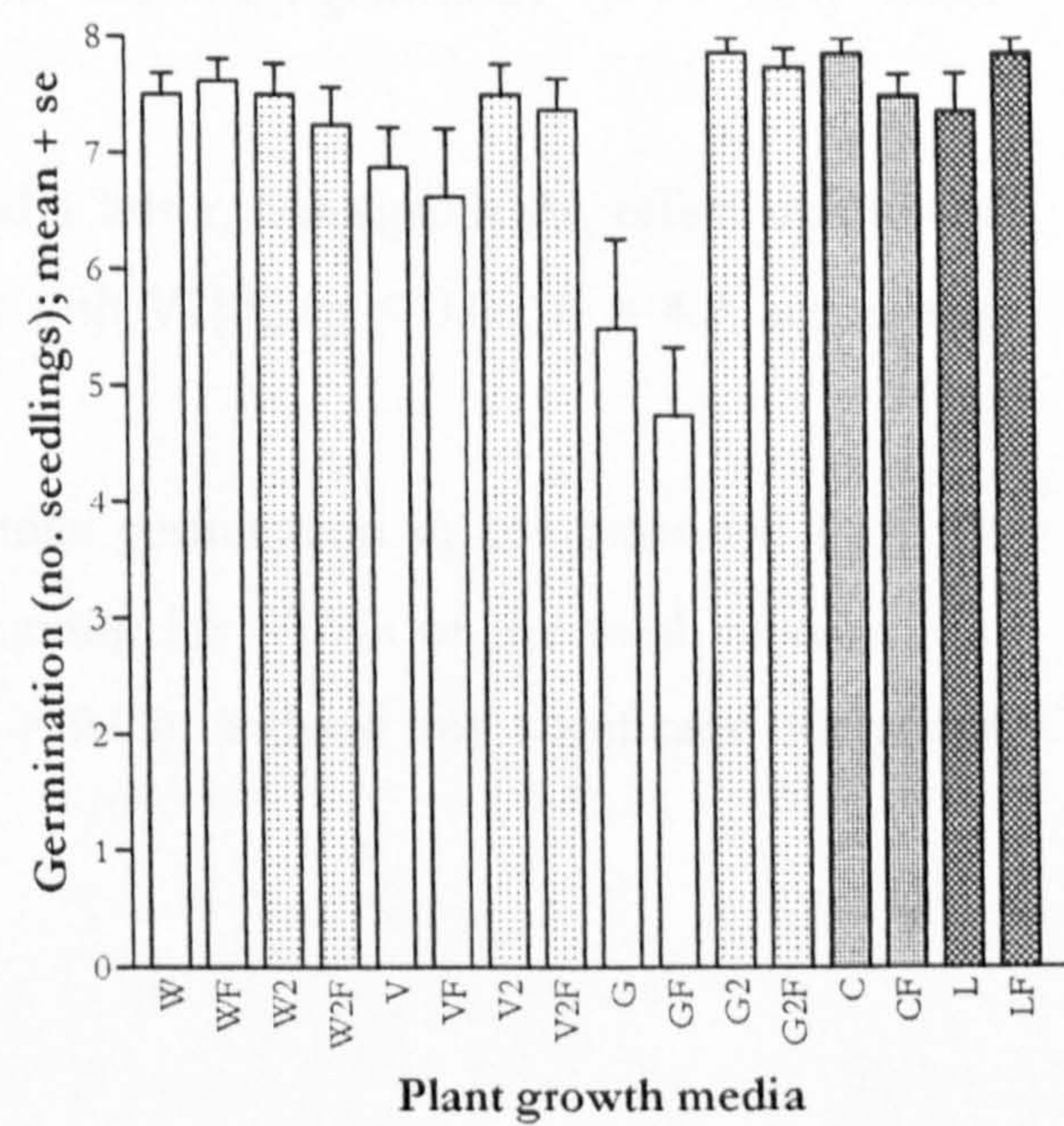


Figure 34 shows large variations in seedling emergence between treatments 7 days after sowing. One-way analysis of variance (one-way ANOVA³) revealed very significant variations between treatments ($p < 0.0001$, $F = 20.4$, $R^2 = 73.2\%$).

Dunnett's multiple comparison test of all treatments against the coir control without fertilization (C), showed significantly ($p < 0.01$) lower levels of germination after 7 days within VCU-WPS (V[F]); GW (G[F]); and coir amended GW without fertilization (G2). It appeared that the amendment of VCU-WPS (V) and GW (G) with coir increased the rate of germination. This was supported by two-way ANOVA; testing both the effect of coir-amendment and fertilization, and any interaction between the two forms of treatment (*cf.* V[F] with V2[F] and *cf.* G[F] with G2[F]). The addition of coir had a very significant effect upon germination within VCU-WPS, $p < 0.0001$, $F = 49.5$, accounting for 69.3% of the total variance; and GW, $p < 0.0001$, $F = 93.4$, accounting for 80.4% of the total variance. The effect of fertilization was not significant ($p > 0.05$) for both VCU-WPS and GW media, with no significant interaction between coir-amendment and fertilization ($p > 0.05$). Neither the addition of coir nor fertiliser had a significant effect on germination ($p > 0.05$) within the W-WPS (*cf.* W[F] with W2[F]); and all W-WPS treatments (W[F] and W2[F]) showed no significant difference from the control ($p < 0.05$).

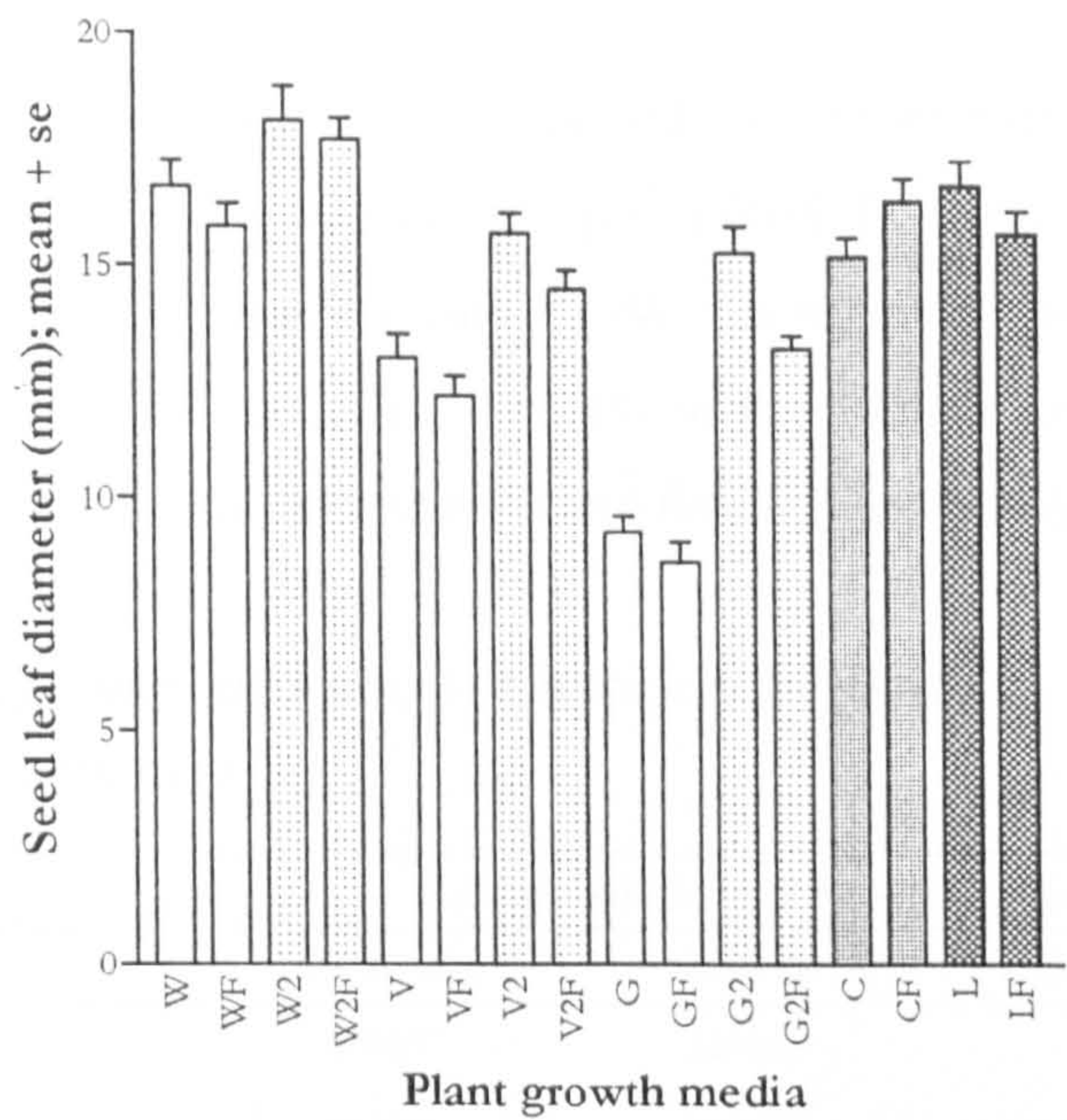
After 14 days, final levels of germination were assessed. Differences in germination were now much smaller, with most treatments achieving $> 82\%$ germination (Table 33). However, one-way ANOVA still showed significant ($p < 0.0001$, $F = 6.2$, $R^2 = 45.3\%$) variations between treatments. Dunnett's multiple comparison tests (using C as the control) now showed that only unmixed GW with or without fertilization showed significantly ($p < 0.01$) lower germination than the control (*cf.* G[F] with C).

Two-way ANOVA revealed coir-amendment had a lesser, but significant, effect upon total germination (14 days) within VCU-WPS (*cf.* V[F] with V2[F]), $p < 0.05$, $F = 4.2$, accounting for 16.4% of the total variance.

GW still showed a very significantly increased total germination in coir-amended form (*cf.* G[F] with G2[F]), $p < 0.0001$, $F = 27.2$, accounting for 55.3% of the total variance. No significant effect of fertilization was shown ($p > 0.05$), without any significant interaction between the two treatments ($p > 0.05$).

³ Bartlett's tests showed standard deviations differed very significantly between treatments ($p < 0.0001$), whereas ANOVA assumes equal standard deviations. However, the Bartlett's test is known to be very sensitive to non-Gaussian data (Motulsky 1990); a Kolmogorov & Smirnov test for normality showed that all data were normally distributed ($p > 0.1$), except ($p < 0.05$) unamended GW with or without fertiliser treatment (G & GF). When G and GF data were excluded from the Bartlett's test, standard deviations were no longer significantly different ($p > 0.05$). ANOVA is known to remain robust when using non-Gaussian data, as long as sample sizes (n) are equal (Motulsky 1990); therefore, the Bartlett's test was ignored.

Figure 36 - Cotyledon diameter (14 days)



Early levels of plant growth, cotyledon diameter after 14 days (Table 33 & Figure 36), varied significantly between treatments, $p < 0.0001$, $F = 29.2$, $R^2 = 79.9\%$; one-way ANOVA.

Dunnett’s multiple comparison test revealed that within all VCU-WPS and GW treatments, cotyledon diameter was significantly ($p < 0.01$) lower than for the fertilised-coir control (*cf.* G[F] and V[F] with CF⁴); with mean differences of 7.1 to 7.7 and 3.3 to 4.1 mm, respectively. Cotyledon diameter was not significantly different for W-WPS and LM treatments ($p > 0.05$).

From Figure 36 it appeared that cotyledon diameter was greater within coir-amended media derived from W-WPS, VCU-WPS and GW; and that fertilization slightly reduced cotyledon diameters in all cases (*cf.* W[F] with W2[F], V[F] with V2[F], and G[F] with G2[F]).

These observations were largely supported by two-way ANOVA, showing significantly greater cotyledon diameters within W-WPS with coir-amendment (*cf.* W[F] with W2[F]), $p < 0.001$, $F = 8.3$, accounting for 26.7% of the total variance. However, fertilization did not have a significant effect ($p > 0.05$), and there was no significant interaction between coir-amendment and fertilization ($p > 0.05$).

Cotyledon diameter within VCU-WPS media was increased more significantly by coir-amendment (*cf.* V[F] with V2[F]), $p < 0.0001$, $F = 32.6$, accounting for 49.2% of the total variance; and significantly reduced by fertilization, $p < 0.05$, $F = 5.5$, accounting for 8.3% of

⁴ The fertilised-coir control was used as nutrient availability would be important to seedling development.

the total variance. There was no significant interaction between coir-amendment and fertilization ($p > 0.05$).

Cotyledon diameter was most significantly increased by coir-amendment within composted green-waste derived media (*cf.* G[F] with G2[F]), $p < 0.0001$, $F = 164.2$, accounting for 80.5% of the total variance. Cotyledon diameter within GW was reduced, also more significantly, by fertilization, $p < 0.01$, $F = 11.0$, accounting for 5.4% of the total variance. Again, there was no significant interaction between coir-amendment and fertilization ($p > 0.05$).

6.3.2 Total plant growth and shoot : hypocotyl ratio

Table 34 - Total radish growth ($\bar{x} \pm se$).

Plant growth media	Fertiliser addition (F)	Individual plant dry mass (mg) after 21 days			S:H ratio :1
		Whole plant	Shoot (S)	Hypocotyl (H)	
VCU-WPS	-	579±57	525±52	55±7	10.0±0.7
Labelled (V)	F	540±60	486±53	55±9	9.5±0.8
VCU-WPS (+50% C)	-	860±67	725±48	135±24	6.4±0.9
Labelled (V2)	F	819±31	717±27	102±9	7.3±0.5
W-WPS	-	981±33	823±23	158±18	5.6±0.5
Labelled (W)	F	1108±65	919±41	189±26	5.4±0.5
W-WPS (+50% C)	-	710±27	531±21	180±24	3.3±0.4
Labelled (W2)	F	1140±48	920±33	220±22	4.4±0.4
Green Waste (GW)	-	229±24	205±21	24±4	9.0±0.8
Labelled (G)	F	149±18	133±17	16±2	8.4±0.7
GW (+50% C)	-	756±57	614±35	143±30	5.3±0.7
Labelled (G2)	F	728±67	623±50	104±20	7.4±1.1
Lev'ton M. P. (LM)	-	1057±90	887±61	170±42	7.5±1.5
Labelled (L)	F	1018±92	874±87	145±30	8.0±1.4
Coir (C)	-	134±07	108±5	27±3	4.2±0.3
Labelled (C)	F	992±81	726±49	267±41	3.0±0.3

Figure 37 - Total plant biomass (21 days)

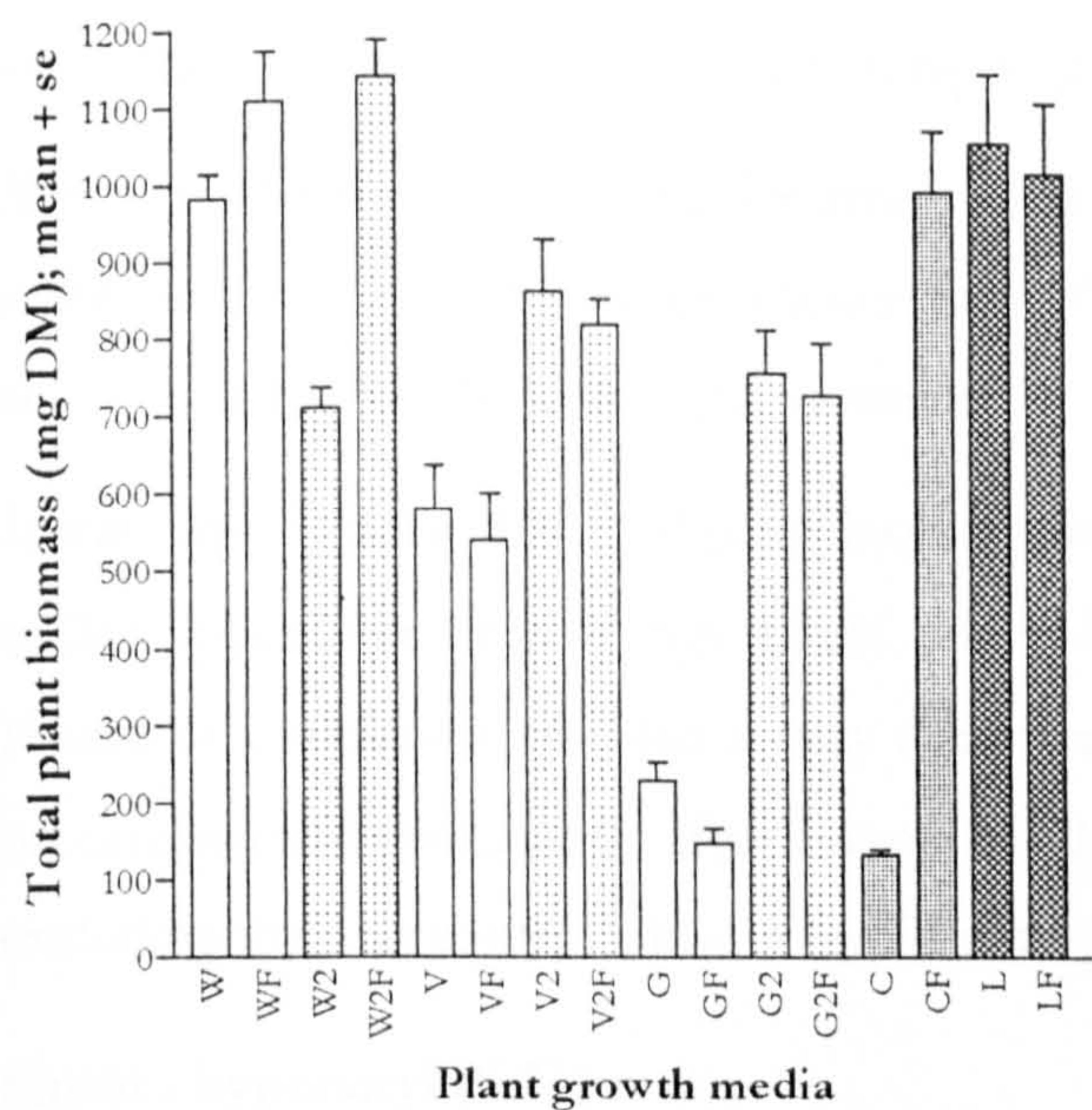
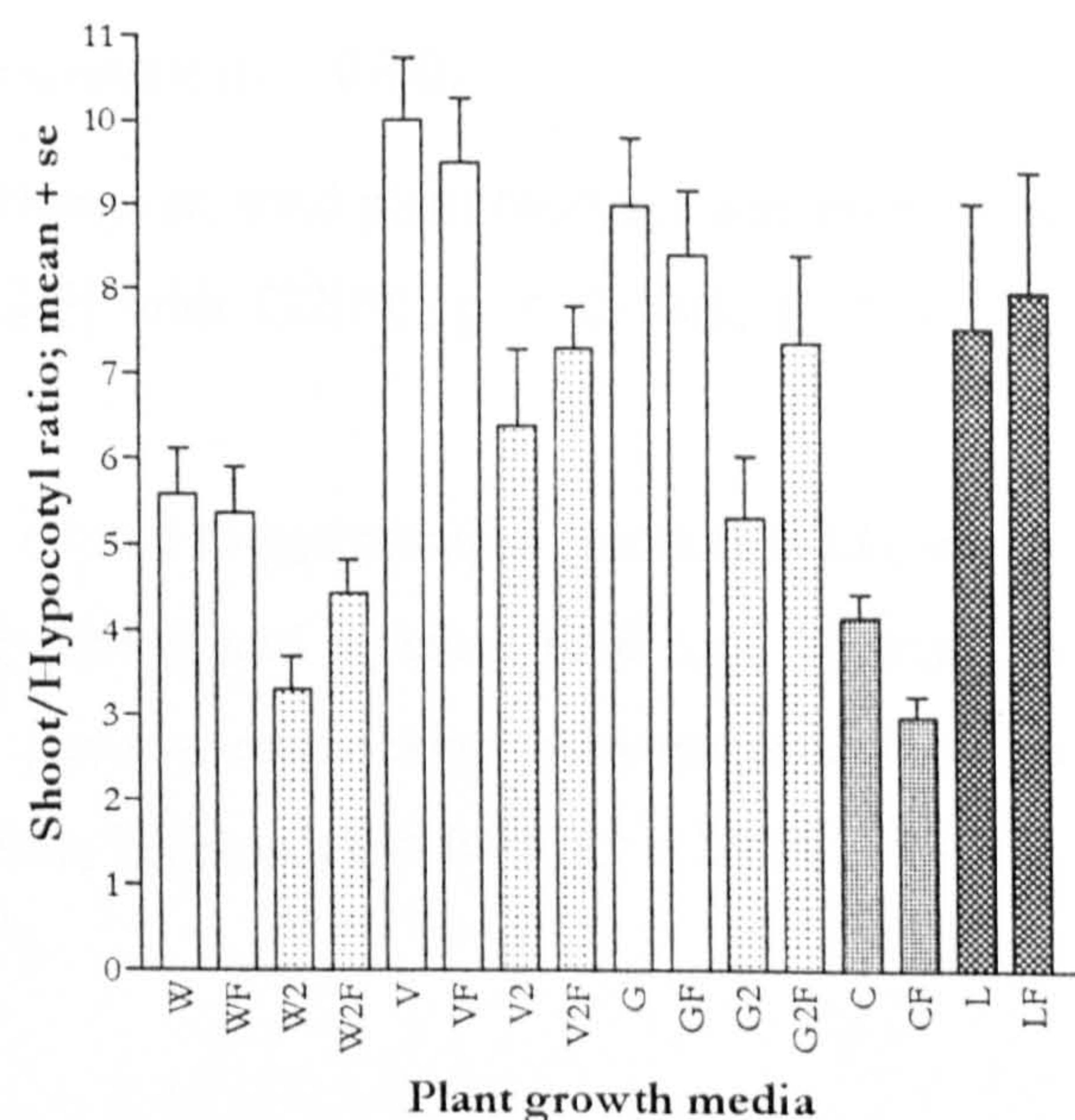


Figure 38 - S:H ratio (21 days)



Total plant biomass production

Significant variation in total plant biomass was observed between treatments ($p < 0.0001$, $F = 33.5$, $R^2 = 81.8\%$; one-way ANOVA).

Total plant biomass achieved within coir-amended and W-WPS with fertilization (W[F] and W2[F]), were significantly higher ($p < 0.05$, Tukey-Kramer’s multiple comparisons test) than those observed for most treatments. Plant biomass was not significantly lower than W[F] and W2[F] in the following treatments: fertilised-coir control (CF); LM with and without fertilization (L[F]); unmixed W-WPS without fertilization (W); and coir-amended VCU-WPS without fertilization (V2).

From Figure 37 it appeared that plant growth within the W-WPS was reduced by the addition of coir (*cf.* W with W2), and greatly increased by subsequent fertilization (*cf.* W2 with W2F). This was supported by two-way ANOVA, which showed total biomass production was significantly affected by the addition of coir (*cf.* W[F] with W2[F]), $p < 0.05$, $F = 6.8$, accounting for 8.2% of the total variance. Fertilization also had a significant effect upon total plant biomass ($p < 0.0001$, $F = 36.9$, accounting for 44.8% of the total variance). An interaction between coir-amendment and fertilization was also detected ($p < 0.01$, $F = 10.9$, accounting for 13.2% of the total variance), showing that fertilization had a greater effect on the increase in total biomass production after the addition of coir.

Low total plant biomass within unmixed VCU-WPS appeared to be increased by the addition of coir, whereas fertilization appeared to decrease biomass slightly in both amended and

unamended VCU-WPS. However, two-way ANOVA revealed that, although coir-amendment had a very significant effect on growth (*cf.* V[F] with V2[F]), $p < 0.0001$, $F = 25.1$, accounting for 46.8% of the total variance, fertilization had no significant effect ($p > 0.05$); with no significant interaction between the two types of treatment ($p > 0.05$).

Similar findings were obtained for green waste. However, total plant biomass was even more significantly increased by coir-amendment (*cf.* G[F] with G2[F]), $p < 0.0001$, $F = 140.7$, accounting for 82.6% of the total variance.

It was hypothesised that total plant biomass was related to germination (section 6.3.1), as the earlier appearance of seedlings would accelerate growth and increase final total biomass. A Pearson's correlation revealed a very significant positive relationship between total biomass production and germination after 7 days ($p < 0.0001$, Pearson's $r = 0.91$, $R^2 = 82.2\%$, $n = 15$), excluding the coir control without fertilization (C⁵).

Shoot : hypocotyl (S:H) ratio

The shoot : hypocotyl ratio (Figure 38), varying biomass partitioning due to environmental stress, revealed significant differences between treatments ($p < 0.0001$, $F = 7.5$, $R^2 = 50.0\%$; one-way ANOVA).

The fertilised-coir control (CF) produced the lowest mean S:H ratio of 3.0 ± 0.3 (Table 34), and largest mean hypocotyl biomass (267 ± 41 mg). Comparing all other treatments with this control (Dunnett's multiple comparison test) revealed greater S:H ratios ($p < 0.05$) within all treatments, except ($p > 0.05$) all W-WPS treatments (W[F] and W2[F]), the coir control without fertilization (C), and coir-amended GW without fertilization (G2).

Two-way ANOVA revealed that S:H ratio was significantly reduced by the coir-amendment of W-WPS (*cf.* W[F] with W2[F]), $p < 0.01$, $F = 12.0$, accounting for 27.9% of the total variance). Whereas, fertilization had no significant effect ($p > 0.05$), and there was no interaction between coir-amendment and fertilization ($p > 0.05$).

Coir-amendment had a similar effect upon the S:H ratio within GW media (*cf.* G[F] with G2[F]), $p < 0.01$, $F = 7.9$, accounting for 20.3% of the total variance; fertilization had no significant effect ($p > 0.05$), and there was no interaction between coir-amendment and fertilization ($p > 0.05$).

The most significant effect of coir-amendment upon S:H ratios was observed for VCU-WPS (*cf.* V[F] with V2[F]), $p < 0.001$, $F = 15.7$, accounting for 35.2% of the total variance. Again,

⁵ Very low levels of total plant biomass production within unfertilised coir was attributed to low nutrient levels.

fertilization did not have a significant effect ($p > 0.05$), and there was no significant interaction between coir-amendment and fertilization ($p > 0.05$).

It was hypothesised that shoot : hypocotyl (S:H) ratio was related to germination. Earlier emergence of seedlings may have indicated more favourable conditions within plant growth media. A Pearson's correlation analysis revealed a significant negative relationship between the variation in S:H ratio and germination after 7 days ($p < 0.05$, Pearson's $r = -0.64$, $R^2 = 40.9$, $n = 16$).

Greater total plant biomass, continued radish growth leading to the convergence of shoot and hypocotyl biomass (Kostka-Rick & Manning 1993), did not explain variation in S:H ratios. A Pearson's correlation analysis of total plant biomass production with S:H ratio, excluding data obtained for the coir control, was not significant ($p > 0.05$, $n = 15$).

6.3.3 Physicochemical evaluation of plant growth media

Table 35 - Physicochemical analyses of experimental plant growth media (determinations were obtained from single composite samples).

Plant growth media	Total nutrients ¹					Water-extractable nutrients ²						
						Macronutrients						
	C	N	C:N	P	K	NH ₄ ⁺	NO ₃ ⁻	P	K	S	Ca	Mg
			:1									
VCU-WPS	34	2.60	13.1	0.90	0.49	30 130	1500 6494	111 481	452 1957	222 961	1434 6208	191 827
50%coir mix	37	2.21	16.7	0.78	0.51	12 69	840 4828	84 483	472 2713	102 586	714 4103	122 701
W-WPS	29	2.25	12.9	0.85	0.52	0 0	132 486	51 187	392 1436	156 571	174 637	27 99
50% coir mix	34	1.76	19.3	0.68	0.53	0 0	6 25	54 229	413 1750	91 386	92 390	22 93
GW	16	0.95	16.8	0.27	1.02	156 292	6 11	48 90	1620 3028	150 280	90 168	18 34
50% coir mix	18	1.00	18.0	0.25	0.99	66 223	9 30	45 152	918 3101	186 628	58 196	14 47
LM	41	1.20	33.3	0.14	0.35	18 120	372 2480	99 660	276 1840	72 480	178 1187	196 1307
Coir (control)	50	0.65	75.4	0.03	0.74	0 0	3 47	5 78	143 2234	96 1500	12 188	4.2 66

Plant growth Media	Water-extractable ²			Mineral macronutrient ratios ³			Ionic factors		Physical factors		
	Micronutrients										
	Fe	Na	Cl	N:P	N:K	K:Mg	pH	EC ⁴	BD ⁵	AFP ⁶	SWC ⁷
VCU-WPS	7 30	203 879	170 736	13.8:1 2.9:1	3.4:1 3.4:1	2.4:1	6.1	2100	615 (231)	27	69
50%coir mix	7 40	248 1425	255 1466	10.1:1 2.8:1	1.8:1 4.3:1	3.9:1	6.3	1410	561 (174)	20	75
W-WPS	7 26	155 568	213 780	2.6:1 2.6:1	0.3:1 4.3:1	14.5:1	7.1	560	563 (273)	28	64
50%coir mix	8 34	203 860	298 1263	0.1:1 2.6:1	0.0:1 3.3:1	18.8:1	7.2	520	507 (236)	19	75
GW	30 56	158 295	1106 2067	3.4:1 3.5:1	0.1:1 0.9:1	90.0:1	8.8	1300	708 (535)	6	49
50%coir mix	23 78	168 568	680 2297	1.7:1 4.0:1	0.1:1 1.0:1	65.6:1	8.6	830	591 (296)	14	68
LM	6 40	67 447	127 847	3.9:1:1 8.6:1	1.4:1 3.4:1	1.4:1	5.5	720	456 (150)	7	78
C (control)	7 109	97 1516	255 3984	0.6:1 21.7:1	0.0:1 0.9:1	34.0:1	6.1	180	440 (64)	22	90

1. All figures = % dry mass.

2. Non-italicised figures = mg l⁻¹ wet mass; italicised figures = mg kg⁻¹ dry mass.

3. Emboldened figures = ratios of water-extractable nutrients available to plant roots (mg l⁻¹); other figures = ratios of total nutrient contents (%DM); N:P ratios for extractable nutrients, N was taken as the sum of NH₄ and NO₃ contents.

4. Electrical-conductivity (µS cm⁻¹) at 25°C.

5. Bulk density (g l⁻¹); figures outside parentheses = wet mass, figures within parentheses = dry mass.

6. Air-filled porosity (% wet mass).

7. Saturated water content (%).

Table 36 - Recommendations, standards and typical values for the physicochemical composition of plant growth media and nutrient solutions (various sources).

Refs.	Macronutrients							
	C	N	C:N ratio	P	K	S	Ca	Mg
1	-	200-250	-	-	-	-	-	-
*	-	60-360	-	26-160	66-400	-	-	-
2	-	200	-	30	200	50	200	40
3	-	-	-	0.001-20	1-50	3-5000	50-1000	1-100
4	2-50	> 0.2	-	> 16	> 121	-	-	> 16
5	> 15/20	< 300	-	< 1200	< 2000	-	-	-
6	-	1.0/1.7	-	0.5/1.0	0.4/0.4	-	-	-
7	-	25-100	-	> 30	90-175	-	> 400	125-300
8	20-38	1.3-2.4	-	0.3-1.6	0.7-1.7	-	3.8-5.4	0.9-2.2
9	-	1.1-4.4	-	0.2-0.4	0.5-1.3	-	-	-
10	19.4	0.6	32.3:1	0.54	0.87	0.39	-	-
*	-	6.7	-	620	1180	144	-	-
11	-	4	-	3-6	460-470	-	57-69	47-56
12	32-35	0.63/0.55	40/46:1	119/100	970/870	13/11	3000/3500	550/540

Refs.	Micronutrients			Mineral macronutrient ratios			Ionic factors	
	Na	Cl	Fe	N:P	N:K	K:Mg	pH	EC
1	-	-	-	-	-	< 3-4:1	4.0-8.0	-
				1.0-2.6:1	0.6-0.9:1	-		
2	20	4	6	7:1	1:1	5:1:1	-	-
3	10-500	10-1000	0.1-25	-	-	-	-	-
4	-	-	-	-	-	-	5.0-8.2	< 2800
5	-	-	-	-	-	-	-	-
6	-	-	-	-	-	-	5.5-8.0	-
7	-	-	-	-	-	-	4.0-6.0	400-850
8	0.3-0.6	-	-	1.3-6.4:1	1.2-2.7:1	0.3-1.1:1	6.5-7.8	610-1370
9	-	-	-	4.3-11.9:1	1.5-3.5:1	-	7.3-8.9	587-1045
10	0.8	-	-	1.1:1	0.7:1	-	8.7	820
*	-	334	-	-	-	-		
11	-	568-774	0.8-0.9	0.01:1	0.006:1	8.4-9.8:1	7.3-7.5	800
12	1300/810	-	3	-	-	0.6:1	8.5	1400

1. Bunt (1976):

recommended content of liquid nutrient media (mg l⁻¹); * composition of added nutrients (mg l⁻¹) during various standard international compost formulations.

2. Hewitt & Smith (1974):

typical concentrations of mineral nutrient solution (mg l⁻¹).

3. Bidwell (1979):

usual levels of water-extractable nutrients in soil solutions (ppm).

4. BSI (1994):

UK standards for topsoils (BS3882); C and N (%DM); extractable P K & Mg (mg l⁻¹); if EC (μS cm⁻¹) > 2800, Na should be determined.

5. Hauke *et al.* (1996):

maximum limits of the extractable nutrient contents (mg l⁻¹) of composted products used as components of plant growth media at ≤40% by volume; C content - minimum requirement of substratum quality compost/matured compost (Germany BKG - RAL quality seal 251); extractable (CaCl₂) N = NH₄+NO₃.

6. Pavan *et al.* (1995):

existing (DPR 915/82)/proposed Italian decree on compost nutrient limits; all figures minima (%DM); except pH (ranged).

7. Verdonk *et al.* (1987):

optimal levels within horticultural substrates for ornamental plants; lower figures for salt-intolerant plants; upper figures for salt-tolerant plants; all figures expressed as mg l⁻¹, except pH (units) and EC (μS cm⁻¹).

8. Nappi & Barberis (1993):

mean nutrient contents of composts derived from various materials (poplar-bark [PB], PB & sewage-sludge, or food industry sludge, and grape stalks & sewage sludge; composted for 8, 4, 4 and 1 month, respectively); all figures %DM, except pH (units) & EC (μS cm⁻¹).

9. TCA (1999):

average nutrient content ranges (min-max) taken from a survey of composts, derived from various feedstocks, currently produced in the UK; N, P, and K (%DM); EC (μS cm⁻¹).

10. Tripepi *et al.* (1996):

paper sludge composted for 39 weeks; total nutrient contents (%DM); * extractable nutrient contents (mg kg⁻¹ DM); pH (units) determined using a saturated paste method; extractable N = NH₄+NO₃ (mg kg⁻¹).

11. Chong & Cline (1994):

composted paper-mill sludge, nutrient contents mg l⁻¹, except pH (units) & EC (μS cm⁻¹); extractable N = NO₃ alone.

12. Hackett *et al.* (1999):

mean nutrient contents of composted pulp, paper mill fly-ash and wastewater treatment sludge (turned windrow/static pile composts, after 34 weeks); N (%DM), other nutrients in available forms (ppm), pH (units), and EC (μS cm⁻¹).

From Table 35 it can be seen that media derived from VCU-WPS and W-WPS were much higher in total nitrogen (N) and phosphorus (P) than the green-waste compost (GW) or Levington's multipurpose compost (LM).

GW derived media contained much higher levels of total potassium in proportion to other macronutrients, giving low total N:K and high K:Mg ratios.

All media contained total C and N within the recommended UK levels for topsoils (Table 36, ref. 4). However, VCU-WPS, W-WPS derived media contained levels of total N and K in excess of Italian limits (Table 36, ref. 6); and GW derived media contained K in excess of Italian limits. However, total NPK contents were broadly comparable to other composts (Table 36, refs. 8 and 9).

The availability of macronutrients to plants (water-extractable nutrients) varied greatly between different media. Readily available forms of nitrogen (water-extractable NH_4 and NO_3) were much higher within unamended VCU-WPS (6624 mg kg^{-1}) and LM media (2600 mg kg^{-1}); equating to 25.5% and 21.7% of their total nitrogen contents (%DM), respectively. In terms of availability to plant roots (mg l^{-1}), VCU-WPS media contained 2 to 4-fold higher levels of water-extractable nitrogen ($852\text{--}1530 \text{ mg l}^{-1}$) than LM commercial medium (non-italic figures - Table 35). This was far in excess of recommended levels (Table 36, refs. 1, 2 and 5).

Within VCU-WPS media, this contributed to higher N:K (1.8:1 to 3.4:1), and much higher N:P ratios (10.1:1 to 13.8:1) compared to all other experimental media (Table 35). These ratios were higher than recommended nutrient levels (Table 36, refs. 1 and 2).

Water-extractable forms of nitrogen within W-WPS compost (132 mg l^{-1}) and GW compost (162 mg l^{-1}) equated to only 2.2% and 3.2% of their total nitrogen contents, respectively. This was reduced further within coir-amended W-WPS (6 mg l^{-1}) to 0.14% of its total nitrogen content; and to a lesser extent, within coir-amended VCU-WPS and GW, (22.2% [852 mg l^{-1}] and 2.5% [75 mg l^{-1}], respectively). This contributed to very low N:P and N:K ratios within the coir-amended W-WPS (0.1:1 and 0.01:1, respectively) compared to recommendations (Table 36, refs. 1 and 2) and the other experimental media (Table 35).

Water-extractable forms of phosphorus were similarly low for W-WPS (51 mg l^{-1}) and GW (48 mg l^{-1}). Extractable phosphorus accounted for 2.2 and 3.3% of their total phosphorus contents, respectively; rising after the addition of coir to 3.4 and 6.1%, respectively. Water-extractable phosphorus within VCU-WPS (111 mg l^{-1}) was > 2-fold higher, accounting for 5.3% of its total P content, which rose to 6.2% after the addition of coir. This was similar to levels of water-extractable phosphorus within LM (99 mg l^{-1}); although LM contained a much higher level in proportion to its total phosphorus content, accounting for 47.1% of its total P content.

Concentrations of water-extractable potassium for unamended VCU-WPS (452 mg l^{-1}) and W-WPS (392 mg l^{-1}) were comparable to other composts (Table 36, refs. 1, 5, 10, 11 and 12), but higher than some recommendations (Table 36, ref. 7) and LM (Table 35). Water-extractable potassium accounted for 39.9% and 27.6% of their total potassium contents, respectively.

High to very high levels of water-extractable K (918 to 1620 mg l^{-1}) were observed for GW media (Table 36, refs. 1, 2, 7, 10, 11 and 12). This contributed to low N:K ratios (0.1:1) within the GW media (cf. Table 36).

Water-extractable calcium was much higher within VCU-WPS media (714 - 1434 mg l^{-1}) than all other media ($< 180 \text{ mg l}^{-1}$), containing at least 4-fold higher levels than W-WPS, GW and LM media (Table 35). These levels were far in excess of typical nutrient solutions (Table 36, ref. 2) and in unamended VCU-WPS, in excess of the normal range for soils (Table 36, ref. 3). However, higher levels have been recorded for other WPS derived composts (Table 36, ref. 12).

Water-extractable magnesium was also much higher within VCU-WPS (191 mg l^{-1}) compared to other W-WPS and GW derived experimental media ($< 30 \text{ mg l}^{-1}$); but similar to the LM media (196 mg l^{-1}) and recommended ranges (Table 36, ref. 7).

Low magnesium contents contributed toward much higher water-extractable K:Mg ratios (mg l^{-1}) within W-WPS and GW media (14.5-18.8:1 and 65.6-90.0:1, respectively). VCU-WPS and LM media contained K:Mg ratios of < 4 , within recommended limits Table 36, ref. 1 & 2).

Water-extractable sulphur and micronutrients within the experimental growth media did not vary as greatly as the other nutrients discussed. However, unamended VCU-WPS, W-WPS, and GW media contained 2 to 3-fold higher levels of extractable S than LM (Table 35). Unamended VCU-WPS, and GW media, contained water-extractable S higher than some WPS composts (Table 36, refs. 10 and 12).

VCU-WPS, W-WPS and GW also contained 2 to 3-fold higher concentrations of sodium (150 - 250 mg l^{-1}) than LM; although these were within the recommended range for soils (Table 36, ref. 3) and below levels observed for composted WPS and other materials (Table 36, refs. 8, 10 and 12).

GW media contained concentrations of water-extractable iron (23 - 30 mg l^{-1}) 3 to 4-fold higher than the other experimental media. These levels are far in excess of normal nutrient solution concentrations (Table 36, ref. 2), other WPS composts (refs. 11 and 12), and are in the region of the highest levels usually recorded for soils (Table 36, ref. 3).

GW media also contained 3 to 9-fold higher levels of chloride ions than the other experimental media. Unamended GW contained chloride in excess of normal soil levels (Table 36, ref. 3).

Electrical conductivity (EC) of the plant growth media solutions revealed that concentrations of total conducting ions were, as expected, greatest within unamended VCU-WPS (1410-2100 $\mu\text{S cm}^{-1}$) and GW (830-1300 $\mu\text{S cm}^{-1}$) composts due to their higher levels of extractable nutrients. A Pearson's correlation analysis of all media showed a very significant correlation between total water-extractable macronutrient/micronutrient contents (mg l^{-1}) and EC ($p < 0.0001$, $R = 0.98$, $R^2 = 95.5\%$, $n = 8$). The EC of VCU-WPS media and unamended GW compost was higher than recommended limits and other composted materials (*cf.* Table 36); and far in excess of the limit of 250 $\mu\text{S cm}^{-1}$ recommended for peat (BSI 1990). The EC of VCU-WPS and GW was reduced after coir-amendment by 32.9% and 36.2%, respectively. This was probably largely due to decreases in bulk density (mg l^{-1} DM) of 24.7 and 44.7%. W-WPS media possessed the lowest EC levels (520-560 $\mu\text{S cm}^{-1}$), mainly a consequence of low water-extractable N. Bulk density and EC within W-WPS were little affected by coir-amendment (decreasing only 13.6 and 7%, respectively).

The pH of experimental plant growth media solution varied considerably. GW media showed the highest pH (8.6-8.8), exceeding most recommendations and typical values (*cf.* Table 36). Both VCU-WPS and W-WPS media possessed lower pH levels (6.1-6.3 and 7.1-7.1, respectively). These pH levels fell within ranges of other composts, but were in excess of some recommended levels (Table 36, ref. 7), and levels (pH 5.0-5.5) thought to be optimal for organic soils and peat in terms of the mineral nutrition of plants (Bunt 1976). All compost-derived media were above the LM medium, which fell within the above recommendations.

Physical factors (*e.g.*, bulk density, air filled porosity and saturated water content) also varied between plant growth media. The air-filled-porosity (AFP) of both the VCU-WPS and W-WPS composts were 27% and 28%, respectively.

Although there is no accepted optimum for AFP, values of between 10-15% are considered desirable (Bunt 1976; Paul & Lee 1976); levels between 9-16% would provide easiest water management in peat, and AFP values of 19-26% may require continuous watering (BSI 1990).

Much lower AFP within the GW or LM composts (6% and 7%, respectively) were within the range of 5-11% recommended for seed composts, but may lead to over-watering (BSI 1990).

The AFP of VCU-WPS and W-WPS was lowered by coir-amendment (20% and 19%, respectively). The AFP of GW was increased > 2-fold by coir-amendment to 14%, as well as increasing its water holding properties; saturated-water-content (SWC) increased from 49% to

68%. SWC was also increased in within VCU-WPS and W-WPS with coir-amendment, to levels similar to LM medium.

Table 37 - Potentially toxic elements (PTEs).

Plant growth media		Units	Atomic symbol								
			Al	As	Cu	Zn	Pb	Ni	Cr	Cd	Hg
Fresh waste paper sludge (WPS) feedstock	Mean \pm se	ppm	-	-	46.4 \pm 1.4	121.1 \pm 4.4	27.5 \pm 2.0	5.2 \pm 0.3	10.0 \pm 0.2	0.6 \pm 0.0	< 0.5
VCU-WPS		ppm	13945	19.3	54.1	271	35.9	9.9	12.0	1.0	< 0.5
		mg l ⁻¹	3216	4.4	12.5	62.5	8.3	2.3	2.8	0.2	458
W-WPS		ppm	20095	22.4	69.4	241.3	45.3	10.1	14.9	0.9	< 0.5
		mg l ⁻¹	5487	6.1	19.0	65.9	12.4	2.8	4.1	0.2	461
GW		ppm	8598	15.3	18.9	116.9	90.7	11.0	13.2	1.8	< 0.5
		mg l ⁻¹	4596	8.2	10.1	62.5	48.5	5.9	7.1	1.0	243
LM		ppm	1276	6.4	67.8	25.5	27.9	1.6	3.2	1.6	< 0.5
		mg l ⁻¹	192	1.0	10.2	3.8	4.2	0.2	0.5	0.2	174
Coir (control)		ppm	831	3.8	9.4	20.7	5.6	1.6	1.6	1.6	< 0.5
		mg l ⁻¹	53	0.2	0.6	1.3	0.4	0.2	0.2	0.2	52
Survey of UK composts, TCA (1999) ¹	Min Max	ppm	-	-	31.6 84.3	117 267	18 127	5.3 25.4	5.5 42.4	0.37 0.86	0.05 0.25
UK and international standards for composts and similar products (maximum contents)											
UK soil association**	Soil Manure	ppm	-	-	50	150	100	50	150	2.0	1.0
			-	-	400	1000	250	100	150	10	2.0
UK OWCA**		ppm	-	15	400	1000	250	100	1000	10	2.0
Austria *	S2200-1	ppm	-	-	70	210	70	42	70	0.7	0.7
	S2200-2		-	-	100	400	150	60	70	1.0	1.0
	S2022 MSW†		-	-	400	1000	500	100	150	4.0	4.0
Germany*	UZ-45	ppm	-	-	75	300	100	-	-	1.0	-
	GZ-251		-	-	100	400	150	50	100	1.5	1.0
Netherlands BRL-K256/02*	HQual-VFG‡	ppm	-	5	25	75	65	10	50	0.7	0.2
	VFG		-	15	60	200	100	20	50	1.0	0.3
Eco-label*	Soil improver	ppm	-	10	100	300	100	50	100	1.0	1.0
Range of limits*	Minimum	ppm	-	5	25	75	65	10	50	0.7	0.2
	Maximum		-	15	400	1000	500	100	1000	10	4.0

1 composts derived from various feedstocks (source-segregated [SS] green waste [GW]; SS vegetable, fruit & garden waste [VFG]; mixtures of GW & VFG; commercially produced compost; and SS materials of undetermined origin)
* figures taken from TCA (1999)
** figures taken from DETR (1999b)
† Municipal solid waste compost
‡ Vegetable, fruit & garden waste compost

Most total metal contents of all plant growth media fell below most suggested limits for potentially toxic elements (Table 37).

However, levels of aluminium (Al), not included in PTE limits, were very high within the W-WPS (approx. 2.01%DM) and VCU-WPS (approx. 1.39%DM); and to a lesser extent GW (approx. 0.86%DM).

Windrow-composted and VCU-WPS compost contained arsenic (As) above the highest suggested legislative limit, and above the other experimental media. Copper (Cu) and zinc (Zn) contents were also in excess of some of the more stringent legislative limits (Netherlands BRL-K256/02, UK Soil Association limits for soils). However, LM, a commercial medium, also contained copper in excess of these limits.

Lead (Pb) contents for all media, except GW, were below all limits.

Nickel contents were also slightly elevated in VCU-WPS, W-WPS and GW composted products, but below all but the most stringent limits (Table 37 - Netherlands BRL-K256/02 for high quality VFG).

The total Chromium (Cr) levels in W-WPS, VCU-WPS and GW, though above levels within LM, were well below all recommended limits.

Cadmium (Cd) levels within VCU-WPS and W-WPS were below that of GW, LM and coir, and all but the more stringent international limits listed.

6.4 Discussion

6.4.1 Plant growth responses

Early (7 days) germination rate

Highest 'early' germination rates, after 7 days, were seen within the coir control (81-83%) and commercial potting medium (83-92%). Although lower rates were seen within the windrow-composted-WPS derived media (63-81%), they were not significantly different from the control ($p > 0.05$, Dunnett's multiple comparison test). This was due to large variations in early germination rates.

Early germination within unamended vermicomposted-WPS (16-25%) and green-compost (2%) showed significantly lower levels of early germination ($p < 0.01$, Dunnett's multiple comparison test). However, germination within coir-amended media derived from these materials increased very significantly ($p < 0.0001$, two-way ANOVA), irrespective of the addition of fertiliser ($p > 0.05$, two-way ANOVA). Windrow-composted-WPS derived media was not significantly affected by coir-amendment or fertilization ($p > 0.05$, two-way ANOVA).

Total (14 days) germination rate

After 14 days, 'total germination' rates were high ($> 90\%$) for most media. Unamended vermicomposted-WPS and green-waste compost derived media still showed lower mean levels of germination (83-86% and 59-69%, respectively). However, significantly lower total germination was observed for the unamended immature green-waste compost ($p < 0.01$, Dunnett's multiple comparison test).

Differences in germination were reflected in significant differences in cotyledon diameter after 14 days. Unamended vermicomposted-WPS and green-waste produced significantly smaller cotyledons than the coir control ($p < 0.05$, Dunnett's multiple comparison test). However, cotyledon size was very significantly greater with the vermicomposted-WPS and green-waste by coir-amendment ($p < 0.0001$, two-way ANOVA); and reduced, less significantly, by fertilization ($p < 0.05$, $p < 0.01$, two-way ANOVA). Cotyledon size within windrow-composted-WPS derived media was also increased significantly by coir-amendment ($p < 0.001$, two-way ANOVA), although it was unaffected by fertilization ($p > 0.05$, two-way ANOVA). Coir-amended windrow-composted-WPS was the only medium showing a significantly greater cotyledon size than the fertilised coir-control ($p < 0.01$, Dunnett's multiple comparison test).

Total plant biomass

Levels of total radish biomass production within the experimental plant cultivation media were broadly comparable to those obtained in previous studies (Kostka-Rick & Manning 1993). However, hypocotyl production was generally low, resulting in high shoot : hypocotyl (S:H) ratios for all media. This was likely due to the relatively high green-house temperatures (25-35°C); optimal day-time temperatures are normally around 25°C (Kostka-Rick & Manning 1993). Shorter photo-periods associated with the month of cultivation (August) may have also contributed to this phenomenon; the most significant variable observed in the data set investigated by Kostka-Rick & Manning (1993).

However, very significant ($p < 0.0001$, one-way ANOVA) levels of variation in germination rates, plant growth, and carbohydrate partitioning (S:H ratios) were observed between different cultivation media.

The highest mean total plant biomass production (dry mass) was observed within the fertilised windrow-composted-WPS derived media (1108-1140 mg); similar ($p > 0.05$, Tukey-Kramer's multiple comparisons test) to that observed for the commercial medium (1018-1057 mg) and fertilised-coir (992 mg). Lower mean biomass production was observed for all green-waste derived media (149-756 mg), and vermicomposted-WPS derived media (540-860 mg), with and without fertilization. All these differences were significant ($p < 0.05$, Tukey-Kramer's multiple comparisons test), except ($p > 0.05$) for coir-amended vermicomposted-WPS without fertilization (819 mg).

Mean total plant biomass production within the windrow-composted-WPS derived media was significantly decreased by coir-amendment ($p < 0.05$, two-way ANOVA). This appeared to increase by fertilization ($p < 0.0001$, two-way ANOVA); especially within the coir-amended windrow-composted-WPS, suggested by a significant interaction between coir-amendment and fertilization ($p < 0.01$, two-way ANOVA). Mean total plant biomass production within the vermicomposted-composted WPS and immature green-waste compost derived media was significantly increased by coir-amendment ($p < 0.0001$, two-way ANOVA); and was unaffected by fertilization ($p > 0.05$, two-way ANOVA).

Shoot : hypocotyl (S:H) ratio

Compared with plants grown in the fertilised-coir control, shoot : hypocotyl ratios were significantly higher for all plants except those from the windrow-composted-WPS derived media ($p > 0.05$, Dunnett's multiple comparison test). S:H ratios were significantly reduced as a result of coir-amendment for windrow-composted-WPS, vermicomposted-WPS, and immature green-waste compost derived media, ($p < 0.01$, two-way ANOVA).

Total plant biomass and low shoot : hypocotyl ratio were significantly correlated to 'early' germination ($p < 0.0001$ and 0.05 , respectively, Pearson's $r = 0.91$ and -0.64 , respectively, and $R^2 = 82.2\%$ and 40.9% , respectively). However, the exclusion of data obtained for unamended vermicomposted-WPS or immature green-waste negated any significant associations of early germination versus total plant biomass and S:H ratio ($p > 0.05$, Pearson's $r = 0.54$ and 0.07 , respectively, $R^2 = 29.0\%$ and 0.0% , respectively). This suggested that the effects of poor early germination upon later plant growth responses were strongest within unamended vermicomposted-WPS and immature green-waste compost.

6.4.2 Physicochemical composition

Generally, vermicomposted-WPS and windrow-composted-WPS contained higher levels of total mineral nutrients, although immature green-waste compost contained the highest levels of total potassium. Water-extractable nutrient contents varied considerably between media. Many of these differences can be accounted for by differences in nutrient losses using different composting techniques. Vermicomposting resulted in lower nutrient losses than windrow-composting during WPS processing (*cf.* Chapter 4 with 5). Windrow-composting can result in large losses of nitrogen through volatilisation as ammonia (Bernal *et al.* 1993; Körner *et al.* 1997; Chapters 2 and 4) and losses of nitrogen and other nutrient via leachate production (Frederickson 1997; Körner *et al.* 1997; Chapters 3,4 and 7).

Differences in water-extractable nutrients, such as, potassium and phosphorus, were also attributable to differing levels of net mineralisation by microbes. This and other potential factors affecting water-extractable nutrient availability, such as mineral and organic associations as well as the influence pH are discussed; as well as reference to the effects of coir-amendment.

Nitrogen

Water-extractable nitrogen with the vermicomposted and windrow-composted-WPS was present mainly in the form of nitrates. Vermicomposted-WPS derived media contained very high levels of water-extractable nitrate characteristic of high nitrification during the vermicomposting process (Edwards & Burrows 1988; Dominguez *et al.* 1997). Water-extractable nitrogen within the immature green-waste equated was largely in the form of ammonium (93.8%) and resulted in a high $\text{NH}_4:\text{NO}_3$ ratio (26.5:1), characteristic of immature composts (Finstein & Miller 1985).

Nitrates are very mobile ions (Beevers & Hageman 1983), therefore, very low levels of NO_3 observed within the coir-amended windrow-composted-WPS may have been due to microbial immobilisation of nitrogen in the presence of a new carbon source (coir) during the preparation, storage and analysis of plant growth media. Large variation in the immobilisation of nitrogen in peat has been observed (Bunt 1976). Immobilisation may have been facilitated by the occurrence of high levels of microbiological activity within the windrow-composted-WPS. The introduction of carbon, especially in water soluble form, is also known to encourage the denitrification of nitrates in soils (McCarty & Bremner 1993).

The availability of nitrate may also be affected by poly-aluminium-chloride (PAC) within the WPS (Appendix 1), a partly cationic polyelectrolyte. The exchange equilibria of NO_3^- and other anions have been investigated for other cationic polyelectrolytes (Rios *et al.* 1994).

Phosphorus

Levels of water-extractable phosphorus within the immature green-waste compost, and vermicomposted/windrow-composted-WPS derived media were low in relation to total phosphorus contents.

Although organic materials, such as peat, do not usually possess the anion exchange properties of soils, containing little aluminium or iron (Bunt 1976), the composts investigated here were shown to contain high levels of aluminium (esp. vermicomposted and windrow-composted-WPS) and iron (esp. immature green-waste compost). Phosphorus ions may be fixed by adsorption upon aluminium and iron oxides or by precipitation to aluminium and iron phosphates (Robson & Pitman 1983).

The high level of aluminium determined for the immature green-waste compost was possibly derived from large quantities of conifers used in the feedstock; some *Pinus* species contain unusually high levels of aluminium - 1000-2000 ppm (Bollard 1983).

Greater concentrations of aluminium within vermicomposted and windrow-composted-WPS were derived from coagulants, mainly aluminium-sulphate [$\text{Al}_2(\text{SO}_4)_3$] (incorrectly referred to as 'alum' - Smethurst 1992) and smaller quantities of the polyelectrolyte poly-aluminium-chloride (PAC), added during the water clarification process (Appendix 1).

Skene *et al.* (1995) observed significant P fixation by alum-treated water-treatment sludge after the addition of phosphorus fertilizer; and Hewitt & Smith (1974) discuss the potential co-precipitation of aluminium hydroxide [a product of the alum flocculation process] and phosphate.

Vermicomposted-WPS derived media contained relatively higher levels of water-extractable phosphorus. Increased levels of available P in vermicompost have been attributed to increased mineralisation of organic forms of P through the activity of earthworms (Ghosh *et al.* 1999). However, Ghosh *et al.* (1999) also attributed the increased availability of P to reductions in fixation with aluminium, iron and calcium.

The availability of P (H_2PO_4^-) is affected greatly by pH values of organic media solutions (Lucas & Davis 1961, cited in Bunt 1976). Although, the adsorption of anions (SO_4^{2-} and H_2PO_4^-) in soils by aluminium and iron oxides decrease with increased pH (Barrow 1970, Bowden *et al.* 1973), aluminium phosphate precipitation is increased with increased pH levels (Bollard 1983). These interactions with aluminium are most significant at pH values of between 4 and 5 (Bollard 1983). The precipitation of phosphate with calcium is more likely to affect P solubility at alkaline pH values (Bohn *et al.* 1979). An increase in alkali conditions (pH 6 to 7) can greatly increase the precipitation of P with Ca ions [also present in high

concentration within vermicomposted-WPS] forming hydroxy-apatite $[\text{Ca}_5(\text{PO}_4)_3\text{OH}]$ (Robson & Pitman 1983). However, the solubility of Ca, Al and Fe phosphates are highest, simultaneously, in slightly acidic to neutral soils (Bohn *et al.* 1979).

Potassium

High levels of water-extractable potassium within the immature green-waste compost was characteristic of green-waste composts (Rainbow & Wilson 1997). Potassium contents of vermicomposted and windrow-composted-WPS were magnitudes lower and did not differ greatly from each other. Levels of water-extractable potassium within plant growth media (excluding the commercial potting medium and coir-control) appeared to be related largely to total potassium content, although it was not possible to test this statistically. A close linear association between total and extractable potassium content was observed for 40 German composts derived from various materials and operations (Körner *et al.* 1997).

Sulphur

Water-extractable sulphur (SO_4^{2-}), usually a highly mobile element (Delwiche 1983), is not as strongly related to changes in pH at pH values above 5.5 (Lucas & Davis 1961, cited in Bunt 1976); and is present at low concentration in the exchange complex (Bidwell 1979). However, SO_4^{2-} may be desorbed from Fe and Al oxides with increased pH (Nodvin *et al.* 1986).

Variations in S availability within media were most likely due to differences in total sulphur contents. Elevated levels of water-extractable S within the WPS composts may have been derived from the addition of alum $[\text{Al}_2(\text{SO}_4)_3]$ during waste water treatment.

Increased levels of water-extractable S within the amended immature green-waste compost was partly attributed to the microbial oxidation of less water-soluble forms of sulphur (*e.g.*, sulphides) into more water-soluble forms (*e.g.*, sulphates). High levels of water-extractable sulphur were present within the coir amendment, and the energy obtained by chemoautotrophic-microorganisms (*e.g.* *Thiobacillus* and filamentous bacteria) during this process is known to be greater than that from the oxidation of ammonium to nitrate (Delwiche 1983). The displacement of water-extractable S within the coir by chloride ions within the immature green-waste compost, was also attributed to elevated water-extractable S within the amended green-waste medium. Adding a large amount of a weakly bonding ion can result in the displacement and release of a small amount of even more strongly bound ions (Bidwell 1979).

Magnesium and calcium

Vermicomposted-WPS derived media contained a higher concentration of water-extractable magnesium and contained a very high concentration of water-extractable calcium ions. The

availability of calcium and magnesium (Ca and Mg) was partly attributed to differences in cation-exchange-capacity (CEC). Ca and Mg are influenced largely by CEC, which is increased by increased pH (Bunt 1976). Therefore, higher levels of water-extractable Ca and Mg were attributed to the lower pH of vermicomposted-WPS. Nutrient leaching during windrow-composting, and higher pH within the windrow-composted-WPS and immature green-waste compost, also attributed to the presence of much lower levels of water-extractable Ca and Mg nutrients.

pH

The vermicomposted-WPS derived media was more acidic (pH 6.1-6.3) than the windrow-composted-WPS derived media (pH 7.1-7.2); and the highly alkali immature green-waste compost derived media (pH 8.6-8.8) was characteristic of green-waste composts (Rainbow & Wilson 1997).

A lowered pH of vermicomposts has been observed for other vermicomposted waste-water sludges (Hartenstein & Hartenstein 1981). Acidification was partly attributed to the very high levels of nitrification within the vermicomposted-WPS; the early stages of this process can produce acidic conditions (The Open University 1996a). Hartenstein & Hartenstein (1981) hypothesised low pH was due to the production of organic acids.

Acidity can have a large influence upon the availability (water extractability) of nutrients affecting adsorption (AEC) and precipitation of anions and cation exchange capacity (CEC).

The availability of cationic micronutrients, such as, Mn, B, Cu, Zn and Mo (not determined) is also strongly affected by pH (Bunt 1976).

Cation exchange capacity (CEC)

The CEC may have also have changed directly upon dilution with coir. Coir has a CEC in the range of between 39-60meq 100g⁻¹ (Evans *et al.* 1996), whereas humus (present in composted materials) has a CEC of around 150-300meq 100g⁻¹ (Bidwell 1979), 200meq 100g⁻¹ (Bunt 1976). However, the CEC of the composts investigated were not determined; and were most likely to be saturated with high levels of Ca, Mg and K ions. Aluminium, having higher bonding energies than Ca and Mg (Bidwell 1979) may also have had an effect.

High levels in water-extractable Ca within vermicomposted-WPS may have affected concentration of water-extractable of Mg ions, through the displacement upon the bonding sites within the cation exchange complex (Bidwell 1979).

Physical structure

Physical properties of experimental plant growth media varied considerably. Air-filled-porosity was much higher within the vermicomposted and windrow-composted-WPS (27-28%) compared to green-waste compost and commercial medium (6-7%). Saturated-water-content (SWC) of most experimental plant growth media were more consistent (64 to 78%), although green-waste compost had the lowest SWC of 49%.

Summary

It was impossible to isolate principal causes for differences in the availability of nutrients within different plant growth media; and was most likely due to a complex interaction of factors (*e.g.*, pH, CEC, anion exchange capacity (AEC), and ion antagonism).

It is also important to point out that during the plant growth period the nature of experimental media may have changed. Compost shrinkage and rewetting properties are important aspects of plant growth media (Valat *et al.* 1991), which may affect AFP and water availability. The compaction of plant growth media during plant growth has been observed for other compost-derived media (Lemaire *et al.* 1985; Lopez-Real *et al.* 1989).

Container depth can alter AFP through changes in water retention (Bunt 1976); as the containers used in this study were only 5 cm in depth, it is likely that AFP would have been minimal as water retention would be at its highest.

The pH of the media may have changed during the experimental period. The uptake of nitrogen as nitrates can increase pH whereas with ammonium uptake the pH may decline (Beevers and Hageman 1983). Microbial activity within the media may also have varied, with the potential for nitrification, and the mineralisation of other nutrients, especially within the immature green-waste compost. The potential for nutrient immobilisation upon the introduction of coir amendment also existed.

6.4.3 The effect of physicochemical composition on plant growth responses

It would appear that variations in plant growth response were due to variations in the physicochemical composition of plant growth media. Pearson correlation analyses were employed to test associations between variations in the physicochemical composition of plant growth media and germination rate, total plant biomass production and S:H ratios.

The effect of physicochemical characteristics on germination

The inhibition of seed germination by high salt concentrations (salinity) has been widely investigated. Wong & Chu (1985) suggest composts with high conductivity and high K, Mg,

Ca and Na contents might exert considerable salt toxicity on seed germination. High sodium chloride (NaCl) concentrations have been shown to inhibit seed germination via decreased imbibition due to osmotic effects (Egan *et al.* 1997). Specific ionic toxicity induced by sodium chloride has been observed in sorghum spp. (Wahid *et al.* 1998) and acacia spp. (Rehman *et al.* 1997); and has caused reduced germination-speed and growth in rice (Khan *et al.* 1997).

Chloride ion concentration [NaCl] inhibited germination by 50% in some moderately salt-tolerant forage legume spp. at an EC of $\geq 1500 \mu\text{S cm}^{-1}$; especially at higher temperatures, *i.e.* 40°C (Esechie 1995). High chloride concentration in domestic waste derived compost has been considered as equally responsible as heavy metals in causing yield reduction in field vegetable crops (Van Assche & Uyttendaele 1982).

Seed germination in species of Broomrape was inhibited by an ammonium concentration of 90 mg l^{-1} (Westwood & Foy 1999). One species of broomrape has been shown to be more sensitive to ammonium-nitrogen, 26-46% germination at 72 mg l^{-1} , than nitrate-nitrogen, no inhibition at 992 mg l^{-1} (Van Hezewijk & Verkleij 1996). The suppression of shoot elongation in germlings of giant witchweed in the presence of ammonium nitrogen has also been observed, whereas nitrate stimulated shoot development (Igbinnosa *et al.* 1996). High ammonium and pH levels may also produce free-ammonia (Bunt 1976), which can inhibit germination (Okuda & Takahashi 1961). Toxic nitrites may also build up under alkali conditions (Bunt 1976). Gasser (1964) found ammonium salts were more damaging than potassium salts to germinating seeds of kale, barely and wheat, suggesting a toxic effect of ammonium (in Bunt 1976).

Significantly reduced early germination rates (7 days) were observed for unamended vermicomposted-WPS and immature green-waste compost derived media only. These were possibly due to ion-specific toxic effects and the osmotic effects of high total water-extractable ion (salt) contents, measured by high electrical conductivity (EC).

Possible causes of germination inhibition and failure within green-waste compost derived media.

To analyse the possible causes of significantly reduced early germination (7 days) within both immature green-waste compost derived media (section 6.4.1), a Pearson correlation was conducted for all media (without fertiliser treatment), excluding data obtained for unamended vermicomposted-WPS⁶. Very significant correlations were observed between germination and concentrations (mg l^{-1}) of NH_4^+ , $\text{NH}_4^+:\text{NO}_3^-$ ratio, K^+ , Fe^{2+} , Cl^- ($p < 0.001$, $r = -0.96$ [NH_4^+]

⁶ Excluding data obtained for unamended vermicomposted-WPS allowed for clearer comparisons between immature green-waste derived media and other media which did not show significantly reduced germination rates (*i.e.* windrow-composted-WPS derived media, coir controls, and commercial potting media).

to -0.99 [Cl⁻], $R^2 = 91.3\%$ [NH₄⁺] to 98.7% [Cl⁻], $n = 7$). Less significant associations were also observed for pH levels ($p < 0.01$, $r = -0.88$, $R^2 = 77.9$, $n = 7$) and SWC (%) ($p < 0.05$, $r = 0.81$, $R^2 = 65.0\%$, $n = 7$).

It was impossible to isolate principal causes of delayed germination observed within media derived from immature green-waste compost. Many of the factors that showed a significant correlation with germination were strongly correlated themselves (*i.e.* correlation coefficient $r = 0.75-0.99$) and, therefore, may have had no direct toxic effect. Furthermore, some factors may have had a toxic effect in combination only, *e.g.*, a high NH₄⁺ concentration and pH can lead to toxicity via free ammonia production (Okuda & Takahasi 1961). High ammonium content within the unamended immature green-waste derived media (156 mg l⁻¹) was a likely cause of delayed and reduced germination (*q.v.* Bunt 1976, Okuda & Takahasi 1961; Igbinnosa *et al.* 1996; Van Hezewijk & Verkleij 1996; Westwood & Foy 1999).

Chloride ions concentrations were also high within the green-waste derived media (68-1106 mg l⁻¹); another potential cause of germination inhibition (*q.v.* Esechie 1995; Van Assche & Uyttebroeck 1982).

High EC and osmotic effects created by high concentrations of the aforementioned ions and potassium may also have contributed to delayed and failed germination rates (*q.v.* Egan *et al.* 1997; Wong & Chu 1985). Although ammonium does not affect EC greatly it does contribute to osmotic pressure or total salt stress of plants (Bunt 1976). Low saturated-water-content of the green-waste compost may have augmented specific toxic ion concentrations and overall osmotic effects.

Greatly reduced concentrations of ammonium, chloride, potassium, EC, and increased SWC would explain the significantly higher germination observed within coir-amended immature green-waste compost (section 6.4.1).

Other toxic substances that had the potential to be present in the immature green-waste compost derived media include fatty acids (Gonzales-Vila *et al.* 1982). Organic acids, such as acetic acid, propionic and butyric acids, have been suggested to account for the toxicity of fresh refuse compost (De Vleeschauwer *et al.* 1981). However, concentrations of these substances were not determined.

Possible causes of germination inhibition within vermicomposted-WPS derived media

To test for possible causes of significantly reduced early germination (7 days) within the unamended vermicomposted-WPS (section 6.4.1), a Pearson correlation was conducted for all media (without fertiliser treatment), excluding data obtained for immature green-waste compost derived media. Significant correlations were observed between germination and

electrical conductivity (EC) ($p < 0.05$, $r = -0.81$, $R^2 = 66.0\%$, $n = 6$); and, more specifically, SO_4^{2-} , Ca^{2+} ions ($p < 0.05$, $r = -0.86$, $R^2 = 73.8$ [SO_4^{2-}] 73.3% [Ca^{2+}], $n = 6$). Surprisingly, a correlation of water-extractable nitrogen [$\text{NH}_4^+ + \text{NO}_3^-$] and germination rate was not quite significant ($p = 0.07$, $r = -0.77$, $R^2 = 59.5$, $n = 6$).

Again, strong correlations were observed between these factors make interpretation of correlation results difficult (*i.e.* EC with Ca^{2+} , $r = 0.99$; Ca^{2+} with SO_4^{2-} , $r = 0.78$; and both EC and Ca^{2+} with [$\text{NH}_4^+ + \text{NO}_3^-$], $r = 0.98$).

Delayed germination rates (7 days) were most likely due to the high EC and osmotic factors affected by high concentrations of Ca and nitrate within the unamended vermicomposted -WPS (*q.v.* Egan *et al.* 1997; Wong & Chu 1985). Although no significant relationship between nitrate and germination was observed, nitrate has a large effect upon the salinity of compost (Bunt 1976). Specific toxicity due to nitrate is less likely than with ammonium (*q.v.* Igbinnosa *et al.* 1996; Van Hezewijk & Verkleij 1996).

Germination within unamended vermicomposted-WPS was less inhibited than in the immature green-waste compost, despite possessing a much higher EC. This suggests that the species of conducting ions at highest concentration were more important to germination inhibition than total levels of conducting ions present.

Greatly reduced concentrations of nitrates, calcium, sulphate and EC would explain the significantly increased germination observed within coir-amended vermicomposted -WPS, ameliorated further by increased SWC (section 6.4.1).

Wong & Chu (1985) observed significant correlations between ethylene dioxide as well as extractable heavy metal (Cd, Cu, Mn, Pb and Zn) contents of compost extracts with the root length of germinated seeds of tomato, cabbage and carrot. However, no significant correlations between total metal concentration and germination were observed ($p > 0.05$).

The effect of physicochemical characteristics on total plant biomass production and Shoot:hypocotyl ratio (plant-stress)

Total plant biomass

Total plant biomass was very significantly ($p < 0.0001$) correlated to early germination rates; a phenomenon observed for other species, *e.g.*, red beet (Benjamin 1987). The significance of the correlation was mainly due to results obtained for unamended VCU-WPS (V[F]) and GW (G[F]) treatments. Excluding these data gave a Pearson's correlation which was no longer significant ($p > 0.05$, Pearson's $r = 0.54$, $R^2 = 29.0\%$, $n = 11$).

Although no significant correlations between plant growth and the physicochemical composition of media were observed, the significance of the relationship between germination and growth was probably not due to the effect of delayed germination alone, but further reductions in plant biomass production affected by similar physicochemical factors. High levels of soluble nutrients (salts) restrict plant growth either by specific ion toxicity or through general salinity effects by reducing the availability of water to plants (Bunt 1976). When using correctly balanced nutrient solutions, radish have been observed to grow maximally at EC levels of between 200 to 400 $\mu\text{S cm}^{-1}$, varying with the season (summer or winter) and culture medium (sand or granulated rockwool) at 25°C (Sonneveld & Vandenbos 1995). EC values for all experimental media investigated during this study were above this range, and would have been increased upon fertilization (Bunt 1976). Windrow-composted-WPS possessed EC values closest to the optima observed by Sonneveld & Vandenbos (1995). Moreover, the mineral nutrient requirements of field-grown radish are low to moderate (Lorenz & Maynard 1988; Nonecke 1989); and an excess supply of nutrients can become toxic, reducing total dry matter production (Moorby & Besford 1983).

Very low total-plant biomass production within unammended immature green-waste compost may have been exaggerated by the presence of water-extractable nitrogen in the form of ammonium. Ammonium can lead to carbohydrate depletion, especially in young plants; it can also depress the uptake of potassium [very high levels within the green-waste compost would have alleviated this effect] and calcium ions (Bunt 1976); and magnesium and nitrate ions (Beevers & Hageman 1983). Ammonium nitrogen is more harmful to *Saintpaulia* than nitrate nitrogen (Kohl *et al.* 1955 in Bunt 1976). Levels of ammonium above 100 ppm [156 mg l^{-1} /292 mg kg^{-1} DM within the unamended immature green-waste compost] at pH 6.6 reduced dry weight production of *Sinapis alba* (Jungk 1968 in Bunt 1976). Holldampf & Barker 1993 attributed increased restrictions in radish growth to ammonium toxicity rather than reduced nutrient (N, P, K and Ca) uptake with increasing ammonium supply (0 to 270 mg l^{-1}), in acid (pH 4.5) soil.

The 'safer' nature of the nitrate form of available nitrogen, which can be stored in plant tissues without detriment to the plant and its assimilation regulated (Beevers & Hageman 1983), may account, in part, for less severe reductions in plant biomass production within the vermicomposted-WPS derived media.

Increased total plant biomass production within windrow-composted-WPS upon fertilization suggested that nutrient (esp. nitrate) concentrations were sub-optimal for radish growth. This was exacerbated by coir-amendment, which significantly reduced total plant biomass production; fertilization restored by fertilization (section 6.4.1).

The availability of phosphorous to plants within all did not appear to be at growth-limiting levels. Concentrations of water-extractable phosphorous within all media (excluding the coir-control) were within normal ranges for plant growth media. Sanchez *et al.* (1991) did not observe significant increases in radish root biomass in Histosols upon phosphorus fertilization when concentrations of water-soluble P were in excess of 13 mg l⁻¹; well below levels recorded for all experimental plant growth media. Histosols are flooded soils rich in organic matter, also known as *peat* and *muck* soils (Bohn *et al.* 1979).

Although higher levels of water-extractable phosphorus were observed for the vermicomposted-WPS, very high levels of nitrate may have had an antagonistic effect upon the uptake of phosphate by plants (Bouma 1983). However, there is evidence to suggest H₂PO₄⁻ absorption by plant roots is unaffected by NO₃⁻ in solution (Robson & Pitman 1983).

Potassium concentrations of all experimental media were greatly in excess of those required by radish; K fertilization of radish grown in Histosols was not required at soil concentrations of 20-102 mg l⁻¹ water-extractable potassium (Sanchez *et al.* 1991).

Potassium concentrations were moderately high within the vermicomposted-WPS and windrow-composted-WPS, but very high within the immature green-waste compost. The antagonistic affects of high levels of potassium can lead to reduced uptake of calcium and magnesium, leading to nutrient deficiencies within plants (Bouma 1983). The K:Mg ratio of immature green-waste compost derived media was 18 to 22-fold higher than recommended to avoid Mg deficiency in plants (Bunt 1976). Although the windrow-composted-WPS derived media possessed a K:Mg ratio 5 to 6-fold higher than recommended to avoid Mg deficiency, this did not significantly restrict total plant biomass production.

Very high levels of calcium ions within the vermicomposted-WPS may have induced reductions in the uptake of manganese, zinc (not determined) and potassium by plant roots (Robson & Pitman 1983). Whereas, within immature green-waste derived media, high levels of water-extractable iron may have led to the decreased uptake of manganese (Robson & Pitman 1983).

High levels of chloride ions within the green-waste compost derived media may have been responsible for plant toxicity. Sodium chloride concentrations of 20mM [710 mg l⁻¹ of Cl⁻] produced symptoms of chloride toxicity within avocado plants, and reduced growth by 60% (Robson & Pitman 1983). Chloride ions have been shown to be approximately twice as toxic as sulphate ions to tomato, bean and lemon (Bunt 1976).

Levels of water-extractable sulphur (sulphate) within the vermicomposted-WPS and green-waste compost derived media were 2 to 3-fold higher than the commercial medium and

quantities usually applied in nutrient solutions. The absorption of molybdenum by tomato plants can be strongly inhibited by sulphate (Robson & Pitman 1983). The uptake of SO_4^{2-} is unaffected by the concentration of other anions, except SeO_4^{2-} (Robson & Pitman 1983).

Physical structure of the media may have also influenced plant growth. The most marked difference was the lower SWC content observed for the green-waste compost, although, in terms of water availability, this may have been counteracted somewhat by its lower AFP. Low SWC indicates a lower water holding capacity which may have increased ionic concentrations within the plant growth medium. Low AFP would have reduce drainage reducing the possibility of nutrient leaching, again maintaining high ion concentrations; the converse would have occurred within the vermicomposted and windrow-composted-WPS.

Although there is no accepted optimum for air-filled-porosity (AFP) of media, values of between 10-15% are considered desirable (Bunt 1976; Paul & Lee 1976), providing the easiest water management for peat (BSI 1990). The addition of coir lowered AFP in vermicomposted and windrow-composted-WPS (19-20%), and raised AFP within the coir-amended green-waste compost derived medium (14%), to levels associated with simpler water management. Whereas continuous watering may be required for AFP levels of between 19% to 26%, over-watering can arise at AFP of 5% to 11% (BSI 1990) making plant management difficult (Bunt 1976).

Very similar AFP and SWC values between vermicomposted and windrow-composted-WPS derived media, suggest that differences in plant growth response were most likely due to mineral nutrient concentrations and potentially toxic substances. Possessing a similar AFP to the commercial potting medium, suggested reduced plant growth within the unamended green-waste was most likely due to nutrient concentrations, augmented by low SWC.

The addition of coir appeared to increase the availability of water within media, either by reducing high AFP or increasing SWC. In some instances coir-amendment greatly reduced nutrient concentration, which may have been detrimental to plant growth.

Shoot:hypocotyl (S:H) ratio (plant-stress)

Shoot:hypocotyl ratio was very significantly ($p < 0.0001$) correlated to early germination rates. Excluding data obtained for unamended VCU-WPS and GW media (V[F] and G[F]), gave a Pearson's correlation that was no longer significant, with practically no variation explained by germination rates ($p > 0.05$, Pearson's $r = -0.07$, $R^2 = 0.0\%$, $n = 12$).

However, S:H ratios were not significantly correlated with total plant biomass. This suggested that S:H ratios revealed 'plant-stress' factors which did not result in decreased plant growth. For example, the commercial potting medium produced plants with a relatively high S:H ratio

(7.5 to 8.0) despite high plant biomass production. Within coir-amended green-waste fertilization increased the S:H ratio by 39.6%, whereas total plant biomass was decreased by only 3.7%.

Again the reasons for differences in S:H ratios would have involved a complex interaction of numerous antagonistic factors. However, in terms of S:H ratios this may be complicated by differences in total plant biomass production. As plant-stress may promote shoot production at the expense of hypocotyl production (Kostka-Rick & Manning 1993), this effect may be exaggerated by increased plant biomass production. Therefore, plant-stress which reduced total plant biomass production may have also reduced S:H ratios. For example, total plant biomass production within unamended vermicomposted-WPS without fertilization was 3.9-fold higher than observed for unamended green-waste compost without fertilization; however, S:H ratio was higher for the vermicomposted-WPS than the green-waste compost (section 6.3.2). Therefore, other factors affecting S:H ratios, such as temperature and photoperiod (Kostka-Rick & Manning 1993), augmented by differences in total plant biomass production, may have masked the effect of physicochemical factors within differing media.

Due to such complications, correlations between physicochemical factors and S:H ratios were not sought. Suffice to say that S:H ratios, in the main, supported differences in plant-stress indicated by differing germination rates and total plant biomass production.

6.4.4 Heavy metals and aluminium

Heavy metals

Levels of arsenic within windrow-composted-WPS and vermicomposted-WPS were above the highest suggested legislative limit. However, although these levels were unlikely to effect radish growth, which can tolerate inorganic As concentrations up to between 150 to 300 mg kg⁻¹ in soil (Simon *et al.* 1998), it may restrict the application of such compost, *e.g.*, use in food plant cultivation.

Higher lead concentrations of the immature green-waste compost were unlikely to have restricted plant growth, although some accumulation may have occurred. It has been observed that Pb uptake in radish is slow compared with cadmium (Cd), and a Pb concentration of 25 ppm in soil did not reduce radish growth; at varying pH values/liming regimes (Han & Lee 1996).

Total nickel concentration were unlikely to effect radish growth as they were below levels tolerate inorganic Ni concentrations up to 100 mg kg⁻¹ in soil (Simon *et al.* 1998); levels thought to be toxic to plants are 20 ppm in extractable form (Bunt 1976).

Cadmium concentrations of ≤ 1.6 ppm were below those though to be toxic for radish. Large levels of Cd uptake have be demonstrated by radish, accumulating mainly in the shoots, but did not effect plant growth at a concentration a 1.52 ppm in soil at varying pH values; increased liming and pH reduced the uptake of Cd (Han & Lee 1996). Concentrations of 96 ppm in soils have been observed to produce a 25% yield decrement within radish plants; resulting accumulations in final hypocotyl (bulb) and shoot Cd concentrations of 21 and 75 ppm, respectively, were observed (Bingham *et al.* 1975).

Aluminium

Webb (1994) reported aluminium contents of between 0.2 and 7.4% for various fresh waste paper sludge types; and suggested such high levels may cause toxicity during certain plant cultivation (in particular, lettuce).

Aluminium present in very high concentrations can be toxic to plants, precipitating around plant roots and interfering with the uptake of iron and calcium; it can also interfere with the availability of phosphorus for plant metabolism (Bidwell 1979). Hewitt & Smith (1974) discuss the potential of co-precipitation at higher pH values by aluminium hydroxide with phosphate; and the potential for severe toxicity at levels of between 10 and 50 ppm; availability being increased by lower pH values. As well as adsorption to Al oxides, P may be precipitated from solution in the form of Al phosphate, a process sensitive to pH levels (Robson & Pitman

1983). In clover, aluminium toxicity can be exacerbated by low phosphate, further augmented by moisture stress (Dodd *et al.* 1992). Aluminium may also build up in plant roots and decrease nitrate uptake and utilisation (Calba & Jaillard 1997; Lidon *et al.* 1998).

The uptake of aluminium would normally be low at the pH levels observed for experimental media in this study, due to the formation of insoluble aluminium-phosphates and hydroxides (*q.v.* Robson & Pitman 1983). However, aluminium uptake and toxicity to plants has been observed at higher pH values (aluminium potentially soluble as the aluminate ion at $\text{pH} > 7.5$) and a lowering of pH can occur in the proximate environment of plant roots (Bollard 1983).

The effects of aluminium may have been alleviated by fulvic and humic acids produced during decomposition of organic matter (Harper *et al.* 1995), substances which can fix other potentially toxic metals through their polyelectrolyte properties (Wood 1996). Poly Aluminium Chloride (PAC) added during waste-water-treatment may also fix some heavy metals (Fe, Mn, Cd, Cr and Pb), increased with increased alkalinity (Malhotra *et al.* 1997); and thus may have mitigated metal concentrations.

High levels of Ca [known to be present in high concentration within VCU-WPS] and/or P, in solution, can mitigate the toxic effects of Al in low concentration, precipitating Al from solutions as hydroxides and phosphates, respectively (Robson & Pitman 1983).

Aluminium can also be beneficial to plant growth, low concentrations (5 ppm) stimulating phosphate uptake (Hewitt & Smith 1974). A low concentration (< 1 ppm) of soluble Al has been shown to improve growth of a number of plant species, although this requires further investigation (Bollard 1983).

Other factors affecting PTE toxicity

Although most countries' standards refer to total metal concentrations, Massiani & Domeizel (1996) argue this is not sufficient as the mobility and availability of such elements depends upon their chemical forms, pH, and organic/inorganic ligand content. Petruzzelli (1996) recognises that although the addition of organic composts to soils may increase total metal content, it may not increase their bioavailability as long as pH levels are not too low. In fact, the addition of organic matter (sewage sludge) may increase the amount of metals (Cd, Zn, Pb, Cu) held by the soil (Petruzzelli *et al.* 1994). The uptake and toxicity of PTEs to plants also varies according to the particular metal element and the plant species involved; again, varying with pH levels and other nutrient contents (Hewitt & Smith 1974; Marschner 1983; Amlinger 1996).

Heavy metals are also essential to plant growth (*e.g.*, Fe, Mn, Zn, Cu, Mo), although they are required at rather low concentrations (Marschner 1983). Common deficiencies in Fe, Zn and

Mn may be due reduced availability at high pH, or, in the case of Cu, high organic matter (Marschner 1983). The availability of heavy metals are affected [similarly to other cationic nutrients] by pH and CEC, *e.g.*, Zn, and organic matter content, *e.g.*, Cd and Pb (Marschner 1983).

The importance of potentially toxic metals within composted products will vary according to their application. Future legislative limits to the metals contents of composts will be based not only on type (mulch, soil improvers or growing media), but whether they are used for the cultivation of food plant species (CEN 1999). Levels of heavy metals in composts applied to agricultural land may accumulate if above those normally associated with soils (Amlinger 1996).

6.5 Conclusions

Windrow-composted-WPS outperformed vermicomposted-WPS and immature green-waste compost for the cultivation of radish, both at 50% concentration in a coir-amended plant growth medium and the unamended form. Marketable levels of nursery-plant growth have been obtained using other WPS-derived windrow-composts, without amendment (Chong & Cline 1994).

Increased radish germination rate, total plant biomass and hypocotyl production within windrow-composted-WPS derived media was attributed mainly to its lower water-extractable nutrient content, and therefore reduced salinity and nutrient toxicity. When windrow-composted-WPS was amended with coir, the lower available nitrogen content greatly reduced total plant biomass, but plant biomass was again found to increase as a result of fertilization. Windrow-composted-WPS derived media performed as well as the coir-control and the commercial potting medium, albeit with a slightly delayed germination under some treatments. Any plant toxicity affected by high aluminium content was not apparent, as observed for broad-bean plants when treated with alum-treated waste-water-sludge (Skene *et al.* 1995).

Delayed germination, reduced total plant biomass and lower hypocotyl production within vermicomposted-WPS derived media was attributed to high salinity (EC) and high water extractable nutrient contents (in particular, NO_3^- , Ca^{2+}).

Delayed and failed germination, reduced total plant biomass and lower hypocotyl production within immature green-waste compost derived media was attributed to high salinity (EC), high water extractable nutrient contents (in particular, NH_4^+ , K^+ , Fe^{2+} , Cl^-). The phytotoxic nature of immature green-waste compost was characteristic of insufficiently stabilised materials, which may also contain toxic organic metabolites produced during composting (Zucconi *et al.* 1981).

These inhibitory factors were alleviated significantly when either vermicomposted-WPS or immature green-waste compost, were amended with 50% coir (by volume). Coir-amendment also increased the water-holding characteristics of all compost-derived media. Coir has long been suggested as a potential alternative for peat (Knight 1991); and has been used successfully in mixtures with perlite and sand for the cultivation of various plant species native to Australia (Offord *et al.* 1998).

It is likely that vermicompost-WPS (and immature green-waste compost) would require increased levels of dilution with other components, such as coir, to produce a suitable potting medium. Subler *et al.* (1998) found that when using vermicomposts as components of plant growth media, concentrations as low as 10-20% produced optimal plant growth responses.

Excessively high salt [nutrient] levels [calcium in particular] in compost-derived media may also be alleviated by deliberate leaching practices (*q.v.* Lopez-Real *et al.* 1989). Further investigations are required into the tailored transformation of composted-WPS into effective plant growth media, as has been conducted for green waste (Rainbow & Wilson 1997).

Although aluminium toxicity was not be evaluated, due to the masking effect of other plant-stress factors, very high total aluminium concentrations in WPS-derived composts suggest further investigation of potential aluminium toxicity would be warranted. In particular, due to increased aluminium solubility under acidic conditions, the application of WPS-derived compost to acid soils or mixing with peat may greatly increase its availability and, therefore, toxic potential. Significant aluminium uptake by subterranean-clover, in mine soil (pH 5.5) after the application of raw paper mill sludge, was observed by Feagley *et al.* (1994). However, the supply of aluminium to tropical acid soil affected by paper sludge application was not documented by Voundi-Nkana *et al.* (1999), although Al toxicity in acid soils was alluded to briefly in introductory remarks. Phillips *et al.* 1997 ignored the potential supply of aluminium to agricultural soils affected by paper mill sludge application. Kraske *et al.* (1993) did not record the availability of aluminium affected by paper mill sludge containing total Al between 26390-31057 ppm, when applied to acid forest soils (pH 4.2-4.5), although the potential mobilisation of Al^{3+} by increased soil solution acidity was implied.

A change in opinion regarding the low potential for aluminium toxicity in ecosystems and animals has been suggested (Gromysz-Kalkowska & Szubartowska 1999). Aluminium is known to be more toxic than Cr^{3+} in some freshwater animals, such as, planarian (Calevro *et al.* 1998); and the effects of aluminium toxicity in soil-animals, such as, earthworms have been investigated (Phillips & Bolger 1998; Rundgren & Nilsson 1997).

Further testing of the WPS derived composts investigated is required. The full assessment plant toxicity requires a suite of test species (OECD 1984), as well as bio-monitoring species to evaluate bio-uptake and bio-accumulation of potentially toxic elements [aluminium, in partular] (Wang 1991). Radish [known to be sensitive to many environmental stresses] has shown signs of aluminium resistance (Zheng *et al.* 1998).

7. A comparison of mechanically turned windrow composting and vermicomposting in the stabilisation of WPS into a horticultural product.

7.1 Introduction

Research into the transformation of organic wastes into potentially valuable components of plant growth media using either traditional composting (Diaz *et al.* 1993), vermicomposting (Edwards & Neuhauser 1988) are well documented. The possible advantages of combining both processes have also been suggested (Frederickson *et al.* 1997; Domiguez *et al.* 1997).

It is recognised that these composting methods operate under very different conditions and can produce very different composted products (Dominguez *et al.* 1997). However, few comparative studies of these processes have been conducted.

Comparisons between traditional composting and vermicomposting focus mostly upon the final composted products (Haimi & Huhta 1987; Subler *et al.* 1998).

Haimi & Huhta (1987) investigated products derived from identical wastes, composted with or without the utilisation of earthworms. However, it was not clear whether composting without earthworms was carried out under optimal conditions, *i.e.* whether adequate aeration or turning regimes were employed. Haimi & Huhta evaluated composts based upon their physicochemical differences, and found it difficult to find significant chemical differences between the composted and vermicomposted organic wastes materials.

Subler *et al.* (1998) also found overlap between the nutritional content of composted and vermicomposted materials, and evaluated different vermicomposted and composted products through plant growth experiments. However, they investigated composts and vermicomposts that were derived from various waste types and obtained from different operations, making direct comparisons unreliable.

There are few comparative investigations into differences between physicochemical transformations during vermicomposting and traditional windrow composting.

Vinceslas-Akpa & Loquet (1997) examined organic matter loss and organic matter transformations during the vermicomposting and composting of maple waste, under small scale laboratory conditions. However, the waste investigated was possibly not suitable for vermicomposting; possessing a C/N ratio of 62/1, the substrate would not have provided

adequate nutrition for earthworms, especially over the 10-month experimental period. The composting aspect of their trial consisted of the incubation of the waste substrate under environmental conditions similar to those used during vermicomposting but without worms. This is not comparable to thermophilic conditions normally associated with large-scale windrow composting systems.

Gellens & Verstraete (1995) evaluated a large-scale batch vermicomposting system, processing pre-composted VFGP waste mixed with non-recyclable paper. The investigation focused upon the processing capacity of the system rather than monitoring changes within organic matter during processing. Also the waste was pre-composted for 3 weeks and reductions in earthworm biomass suggested this substrate offered little nutrition to the earthworm population used.

Traditional composting systems (Campbell *et al.* 1991; Brouillette *et al.* 1996; Jackson & Line 1997a, b, c & 1998) and vermicomposting systems (Elvira *et al.* 1995a, b, 1997 & 1998) have been demonstrated as feasible methods for treating waste paper sludge; and numerous studies into the horticultural and agricultural value of composted and non-composted waste paper sludge have been conducted (Bellamy *et al.* 1995; Campbell *et al.* 1995; Chong & Cline 1993 & 1994; Chong & Hamersma 1996; Tripepi *et al.* 1996).

Waste paper sludge (WPS), amenable to treatment via traditional composting and vermicomposting, with the potential to produce composted products with horticultural value, was an appropriate substrate to use during a comparative study of these two processes, and their respective products.

The WPS investigated in following comparative study was obtained from a packaging recycling mill, an intimate mix of primary and secondary waste-water treatment sludges (Appendix 1). The WPS, containing high levels of organic matter and appropriate levels of mineral nutrients, was suitable for both large-scale windrow composting (Chapter 4) and vermicomposting (Chapters 3 and 5) without nutritional amendment (*cf.* Elvira *et al.* 1997; Jackson & Line 1997a & b).

The following Chapter makes direct comparisons between the open-air mechanically turned windrow composting system (Chapter 4) with the enclosed batch vermicomposting system (Chapter 5) during the processing of waste paper sludge. The performance of each system was evaluated in terms of process (*i.e.*, analysing physicochemical changes in the WPS during processing) and its effect upon the performance of final composted products (*i.e.*, as cultivation plant media).

To achieve a balanced comparison between windrow composting and vermicomposting, optimal results from each trial were drawn. Results for the windrow composting aspect of the study were taken from WPS composted with a wood-chip bulking agent (Chapter 4); whereas results taken from single-batch vermicomposting treatment (Chapter 5) were used for the vermicomposting aspect of the comparison. These two treatments were considered to have provided the most effective composting method in each case.

The performance of the respective composted products was tested using radish plant-growth trials (Chapter 6). As the effects of the physicochemical composition of windrow compost and vermicompost upon plant growth were covered in detail in Chapter 6, only the more salient differences are outlined here.

Materials and methods of windrow composting, vermicomposting and plant growth trials have already been covered in the Chapters 4, 5 and 6, respectively. Therefore, the results only are documented and discussed.

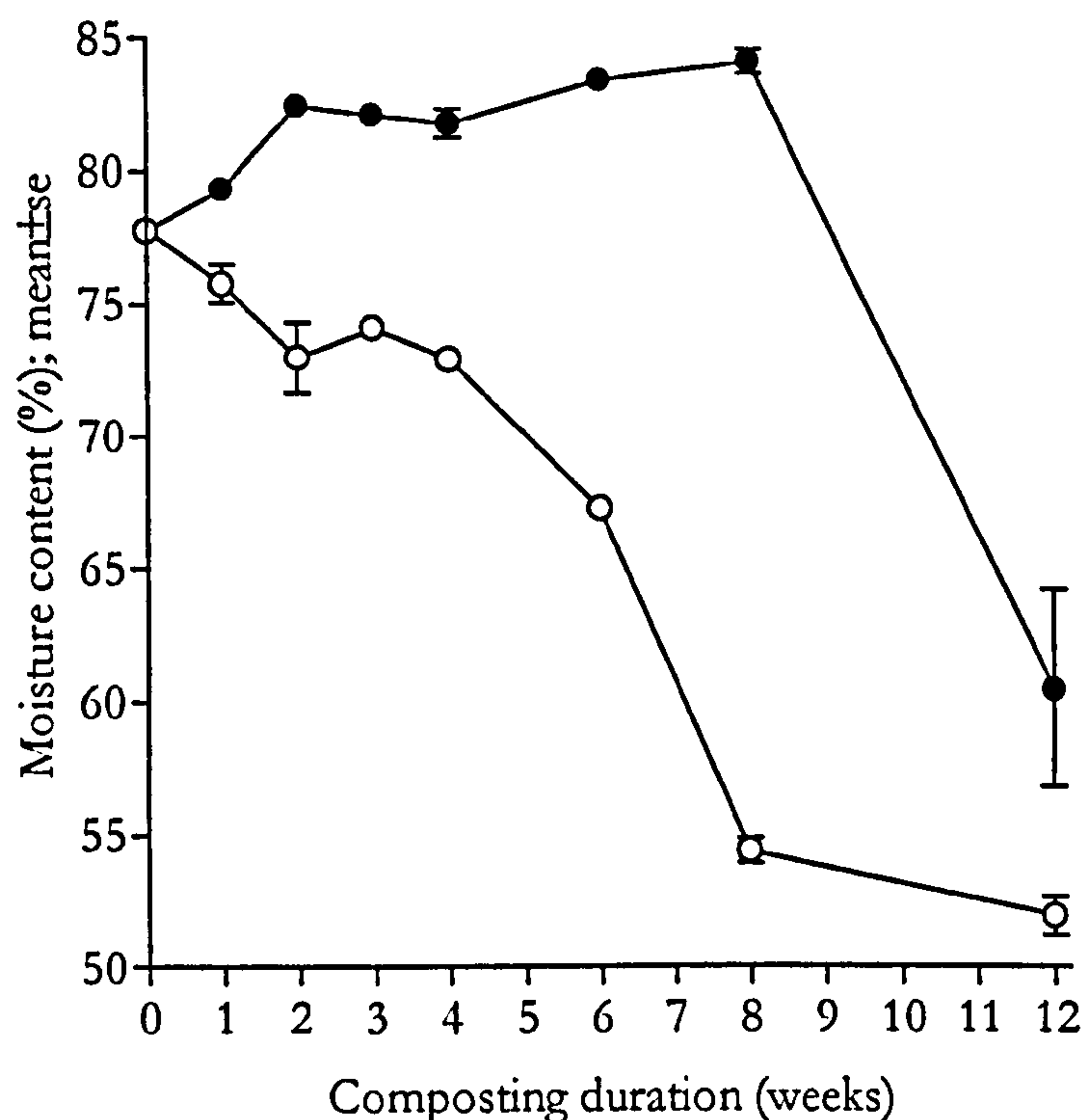
7.2 Results

7.2.1 Comparison of the composting processes.

Moisture

Figure 39 – WPS moisture content

Windrow-composting with bulking agent [—○—]; Single-batch vermicomposting [—●—].



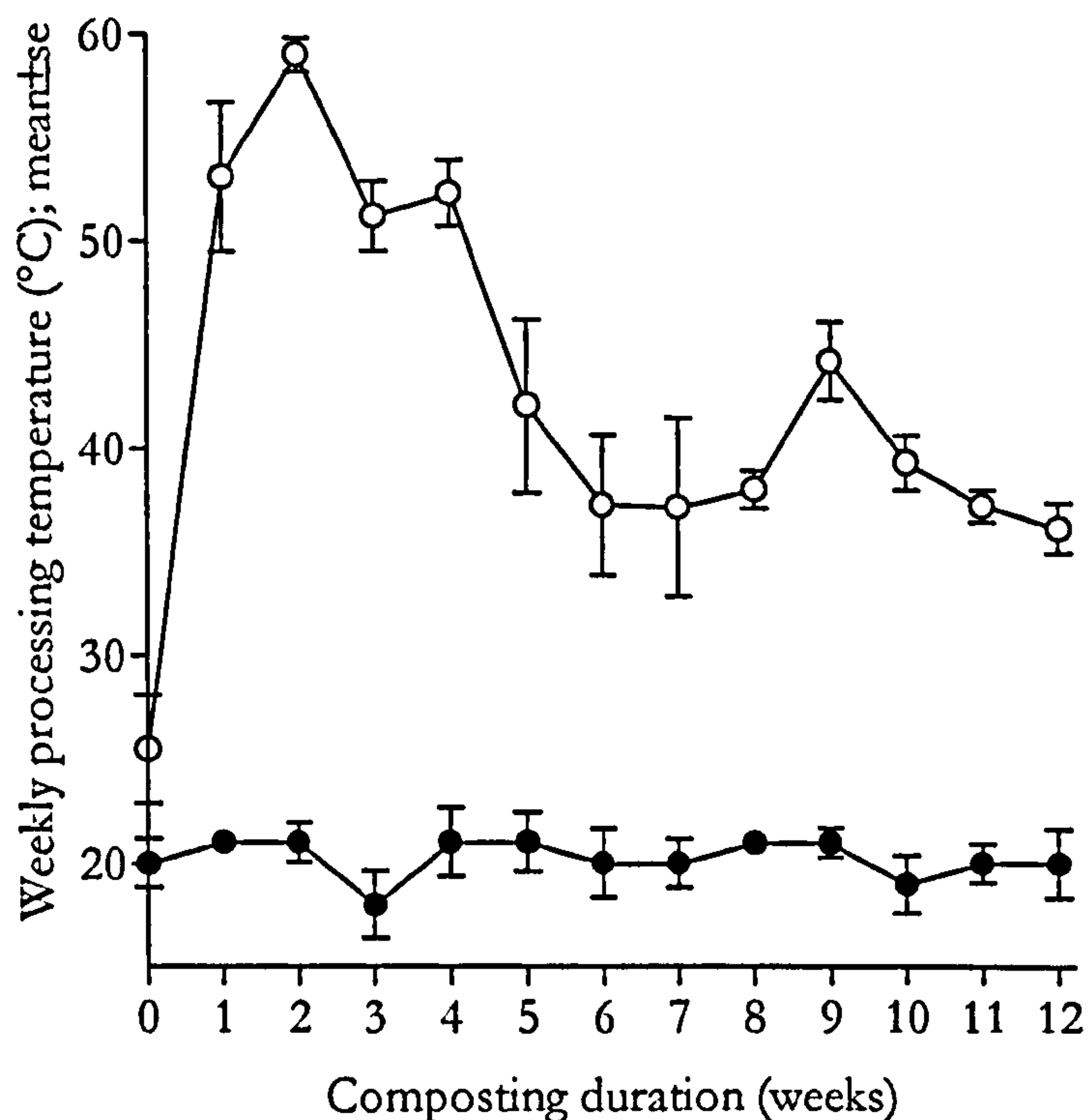
Initial moisture content of the WPS was similar for both the windrow composting and vermicomposting feedstock (78%). However, during windrow composting, moisture decreased rapidly reaching 55% by week 8. During vermicomposting moisture content of WPS rose to 84% by week 8. The moisture content of WPS during windrow composting was significantly below that for the vermicomposting from week 1 to 8 inclusive ($p < 0.01$, Student *t*-test). After maturation (week 12) there was no statistical difference in moisture content ($p > 0.05$); $51.9 \pm 0.7\%$ for windrow composted WPS and $60.5 \pm 3.5\%$ for vermicomposted WPS. The dramatic fall in moisture for vermicomposted WPS between weeks 8 and 12 was achieved by removing the vermicompost from vermicomposting units and maturing it in small piles; a deliberate attempt to accelerate its desiccation before plant growth trials.

However, large variations in moisture content within the vermicompost remained. One sample possessed 49.5% moisture while 3 samples had moisture contents of between 63.5% and 65.5%. Excluding the low moisture reading (Grubb's test for outliers – Motulsky 1995) resulted in a final moisture content of $64.2 \pm 0.7\%$ for the vermicomposted WPS, significantly higher than the windrow composted WPS ($p < 0.0001$).

Temperature

Figure 40 – Processing temperature

Windrow-composting with bulking agent [—○—]; Single-batch vermicomposting [—●—].



Temperatures within the windrow composting WPS were very significantly ($p < 0.001$) higher than those within the vermicomposting system throughout the active composting (weeks 1 to 8), and maturation periods (weeks 9 to 12).

The greatest temperatures during windrow composting were observed at weeks 1 to 4, with mean temperatures of between 51 to 59°C. Differences between windrow and vermicomposting process temperatures during this period, were between 30.7 to 37.9°C. Thereafter, the windrow processing temperature fell to mean temperatures of 36 to 45°C, between 15.5 to 23.1°C higher than the vermicomposting process.

WPS processing temperatures during vermicomposting were maintained at a mean of 20.8°C during the whole vermicomposting period. However, despite thermostatically controlled heating, temperature fluctuations were observed, *e.g.* week 3 showed a mean temperature of 18°C.

Organic matter losses

Total organic matter

Figure 41– TOM of WPS

Windrow-composting with bulking agent [●]; Single-batch vermicomposting [○];
Curves of best-fit ($y=a \times e^{(-\lambda x)} + c$).

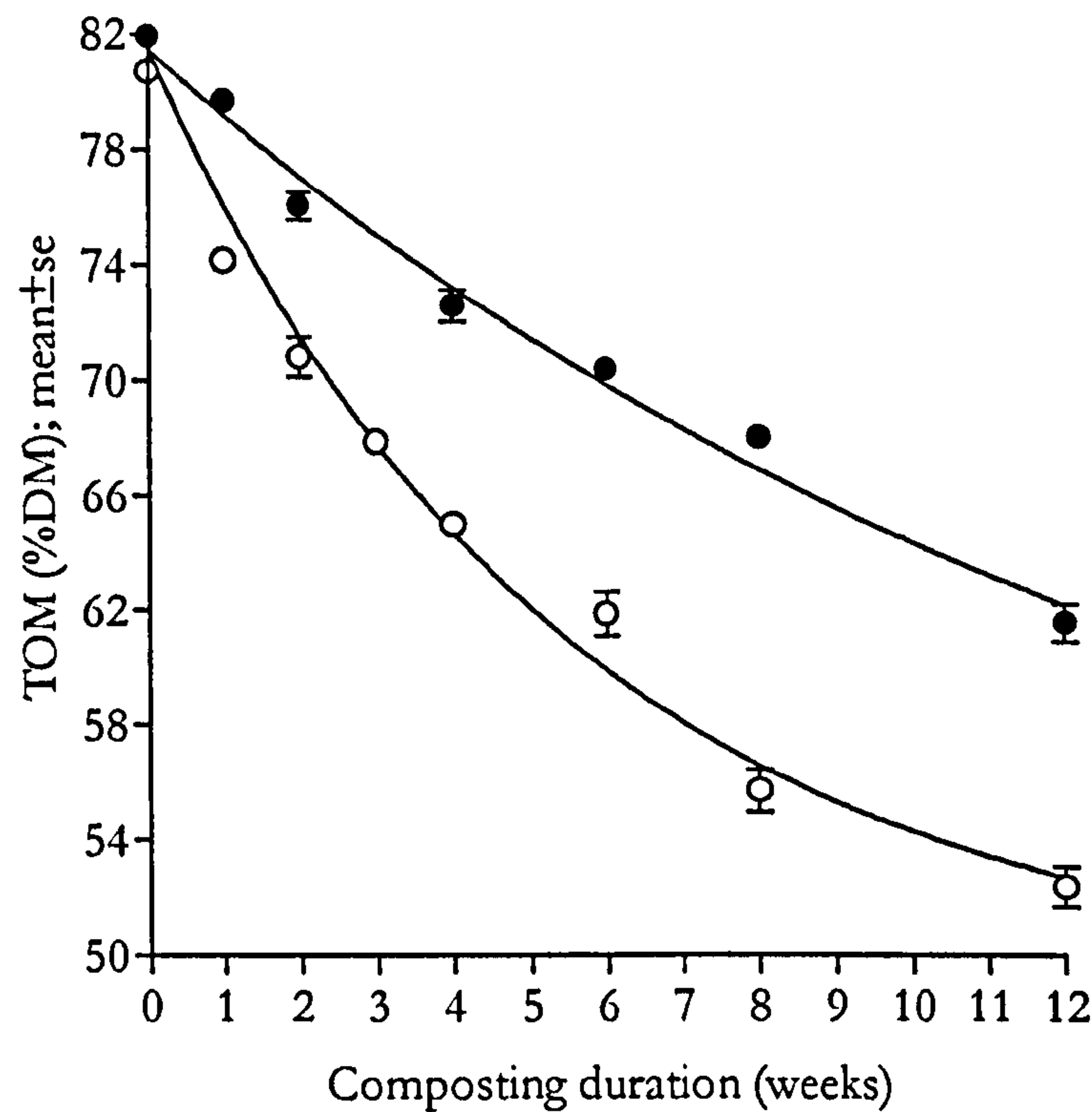


Table 38 – Non-linear regression ($a \times e^{(-\lambda x)} + c$): changing TOM (%DM); mean±se.

Variable	Windrow composting	VCU composting
Total TOM loss (a)	32.79±1.63 ^a	32.17±2.38 ^a
Final TOM (c)	47.07±1.80 ^a	49.85±2.48 ^a
Decay constant (λ)	0.152±0.017 ^a	0.080±0.018 ^b
Half-life (T) weeks	4.57±0.51 ^a	8.69±1.05 ^b
Goodness of fit		
R ²	98.0%	97.8%
S _{xy} (TOM %DM)	1.33	1.02

Abc - different letters indicate significant differences (p<0.05; unpaired Student's t-test).
R² - fraction of total variance in experimental data explained by the model.
S_{xy} - standard deviation of points from the model line-of-best-fit (Motulsky 1996)

High R² and low S_{xy} suggest the non-linear model provided a good fit for the experimental data.

Non-linear regression (Table 38) predicts that windrow composting and vermicomposting may result in similar total TOM losses and final TOM levels (P>0.05), 47.1 and 49.9%DM, respectively.

However, rates of TOM loss was significantly ($p<0.05$, Tukey-Kramer's test) greater during the windrow composting of WPS. The decay constant within the windrow composted WPS (0.152) was 1.9-fold higher than for vermicomposting WPS (0.080), resulting in a 47% shorter half-life (cf. 4.6 weeks for windrow composting with 8.7 weeks for vermicomposting).

This predicts that windrow composting resulted in a TOM loss of 23.53%DM 4.6 weeks, equivalent to a rate of 5.15%DM week⁻¹. Whereas, vermicomposting resulted in a TOM loss of 24.93%DM 8.7 weeks, equivalent to a lower rate of 2.87%DM week⁻¹.

By week 12 actual TOM values of the windrow composted and VCU composted WPS were 52.4±0.7%DM and 61.6±0.3%DM, respectively; with a mean difference of 9.2%DM ($p<0.0001$).

After 12 weeks the windrow composted WPS was 5.3%DM higher ($p<0.05$; unpaired t-test with Welch correction) than the final TOM content predicted by the model ($c=47.1\pm1.8\%$ DM). The final VCU composted WPS was 11.8%DM higher ($p<0.01$; unpaired t-test with Welch correction) than the final TOM content predicted by non-linear regression ($c=49.9\pm2.5\%$ DM).

The larger actual final TOM content, and the larger difference between predicted (non-linear regression) and actual final TOM contents within the vermicomposted WPS (after 12 weeks) suggested a greater level of stabilisation had occurred within the windrow composted WPS.

Greater TOM losses during windrow composting were demonstrated by significantly greater losses in TOM after correcting for loss of dry mass (%ash). During windrow composting very significantly greater TOM (%ash) losses were observed within the WPS at all sampling periods (Table 39).

Table 39 – Loss of TOM based on ash (%ash); mean±se

Sampling period (week)	windrow	vermicompost	Mean difference
1.0	32.6±1.2	7.7±0.3	24.9***
2.0	42.9±1.9	25.4±1.0	17.5***
3.0	50.4±0.1	-	-
4.0	56.4±0.5	37.8±0.9	18.6***
6.0	61.8±1.2	45.3±1.1	16.5***
8.0	70.4±0.9	52.7±2.8	17.7***
12.0	74.1±0.7	62.3±0.5	11.8***

*** $p<0.001$ - unpaired Student's t-test

Mean differences showed the largest difference during the first week processing, diminishing to a final difference of 11.8%ash at week 12 ($p<0.001$).

Table 40 – Fibre analysis of fresh and composted WPS (8 weeks) (mean±se)*

Fibre fraction	Fibre content (%DM)			Fibre losses (%ash)	
	Fresh WPS	Windrow	VCU	Windrow	VCU
Total Fibre (NDF-ash)	53.6±0.6 ^a	37.2±0.7 ^b	43.8±1.3 ^c	70.2±2.4 ^a	53.5±2.8 ^b
Fibre components (%DM)			Fibre losses (%DM)		
Hemicellulose (NDF-ADF)	13.5±0.3 ^a	7.6±0.5 ^b	12.2±0.2 ^a	5.9±0.5 ^a	1.3±0.1 ^b
Cellulose (ADF-lignin-ash)	31.9±0.4 ^a	18.7±0.4 ^b	21.1±0.8 ^c	13.2±0.4 ^a	10.8±0.5 ^b
Lignin (ADF-cellulose-ash)	8.2±0.3 ^a	11.0±0.5 ^a	10.5±0.9 ^a	-2.8±0.2 ^a	-2.3±0.3 ^a

* Fibre determinations were conducted by the Agricultural Development and Advisory Service (ADAS[®]) laboratories, Wolverhampton, UK; NDF – neutral detergent fibre, ADF – acid detergent fibre (Anon. 1986).

^{abc} different letters indicate significant differences between adjacent columns marked with similar units of measurement (p<0.05, Tukey–Kramer post ANOVA multiple comparisons test).

Losses in TOM were reflected in significant changes in total fibre, hemicellulose and cellulose content (Table 40) were seen in both windrow composted and vermicomposted WPS after 8 weeks (p<0.001, Tukey-Kramer test). Correcting for loss of dry mass revealed that total fibre losses (%ash) after 8 weeks were very similar to TOM losses (*q.v.* Table 39) (p>0.05, Student t-test).

Losses in hemicellulose and cellulose contents were 5.9%DM (a 43.7% decrease) and 13.2%DM (a 41.4% decrease), respectively, for windrow composted WPS. These losses were significantly higher (p<0.001, Tukey-Kramer test) than for vermicomposted WPS, which showed a 1.3%DM (a 9.6% decrease) in hemicellulose and 10.8%DM (a 33.9% decrease) in cellulose.

The largest difference in fibre losses between windrow composting and vermicomposting was observed for hemicellulose, 4.6%DM (a 3.5-fold greater loss during windrow composting). The difference between cellulose losses was 2.4%DM (a 22% greater loss during windrow composting).

No significant change was observed for lignin content in terms of %DM, and lignin contents increased (2.3 to 2.8%DM), concentrated by the loss of the other fibre components. Once lignin loss was adjusted, correcting for loss of dry mass (%ash), significantly greater losses (p<0.05, Student t-test) were calculated for windrow composted WPS (42.2%ash) compared to vermicomposted WPS (26.6%ash). Differences in lignin losses (based on ash - %ash) were significantly different (p<0.01, Student t-test).

Changes in nitrogen content

Table 41 – Changes in nitrogen content, in varying forms*

Composting method	Sampling period (week)	TKN %	N loss %ash	TOC %	C/N ratio	Water extractable nitrogen		
						NH ₄ ⁺ mg kg ⁻¹	NO ₃ ⁻ mg kg ⁻¹	NH ₄ /NO ₃ ratio
Fresh	0	2.70±0.06	n/a	44.6±0.1	16.5±0.5	686±78	19±1	36±6
Windrow	2	1.75±0.05	57.7±1.4	39.0±0.4	22.2±0.8	3269±441	23±2	144±38
	4	1.81±0.06	63.7±1.7	35.8±0.1	19.8±0.7	2780±143	20±1	141±16
	6	1.79±0.06	67.0±1.8	34.0±0.4	19.0±0.9	151±41	16±1	10±4
	8	1.89±0.05	70.0±1.8	30.7±0.4	16.3±0.7	213±55	113±51	1.9±1.3
	12	2.13±0.07	68.5±1.9	28.8±0.4	13.5±0.6	6±6	308±48	0.02±0.00
VCU	2	3.58±0.09	-5.4±0.1	41.8±0.1	11.7±0.3	2147±150	26±3	82±17
	4	3.42±0.07	12.2±0.2	39.9±0.2	11.7±0.3	4167±701	28±5	148±51
	6	2.54±0.25	40.3±2.4	38.5±0.4	15.1±1.8	1185±77	45±14	26±10
	8	2.78±0.11	41.3±1.9	36.7±0.8	13.2±0.8	417±39	2656±305	0.16±0.03
	12	2.47±0.04	54.7±1.0	33.9±0.2	13.7±0.3	357±9	6111±230	0.06±0.00

*All figures expressed in terms of dry mass where applicable.

Windrow composting resulted in a significantly greater loss of nitrogen (%ash) than was observed for the vermicomposted WPS ($p<0.001$, Student t-test), at all sampling periods; resulting in higher TKN contents (%DM).

At week 2, nitrogen loss within the windrow composting WPS was 63.7%ash ($p<0.01$, Tukey-Kramer test). Although, no significant difference was observed for consecutive sampling periods, thereafter, a final loss of 68.5%ash (week 12) was significantly higher than observed at week 2 ($p<0.01$, Tukey-Kramer test).

At week 2 the nitrogen content within the vermicomposting WPS increased slightly (5.4%ash, $p<0.01$, Student t-test). Nitrogen losses between consecutive sampling periods were significant ($p<0.001$, Tukey-Kramer test); with a final loss of 54.7%ash (week 12 – 8 weeks composting and 4 weeks maturation).

Final nitrogen losses were significantly different between windrow composted and vermicomposted WPS ($p<0.001$, Student t-test). These losses can be calculated as 18.5g kg⁻¹ WPS and 14.8g kg⁻¹ WPS for windrow composting and vermicomposting, respectively.

C/N ratios within the windrow composted WPS were significantly higher than fresh WPS for weeks 2 ($p<0.001$, Tukey-Kramer test), indicating that nitrogen losses were in excess of carbon losses.

Within the vermicomposted WPS the converse was observed; significantly lower C/N ratios at weeks 2 and 4 ($p<0.01$, Tukey-Kramer test) indicated carbon losses exceeded nitrogen losses.

C/N ratios within the windrow composted and vermicomposted WPS showed similar final values ($p>0.05$, Student t-test) of 13.5 and 13.7, respectively; both significantly below fresh WPS ($p<0.001$, Tukey-Kramer test).

This suggested that proportions of carbon and nitrogen lost overall, during both composting processes, were similar; although greater losses of both elements occurred during windrow composting.

Both composting processes resulted in high levels of water extractable ammonium [NH_4^+] produced during the early stages of composting (weeks 2 and 4), significantly above levels in fresh WPS ($p < 0.01$, Tukey-Kramer test).

NH_4^+ levels fell more rapidly within the windrow composting WPS, showing significantly lower values than the vermicomposting WPS at weeks 6, 8 and 12 ($p < 0.05$, Student t-test).

Significant increases in nitrate [NO_3^-] (2656 mg kg^{-1}) was observed at week 8 for vermicomposted WPS ($p < 0.01$, Student t-test with Welch correction). Although the windrow composted WPS contained increased nitrate (113 mg kg^{-1}), this was not statistically significant ($p > 0.05$, Student t-test with Welch correction) due to a large level of variation.

The nitrate content of the windrow composted (308 mg kg^{-1}) and vermicomposted WPS (6111 mg kg^{-1}) after 12 weeks were both significantly higher than fresh WPS (19 mg kg^{-1}) ($p < 0.001$, Student t-test with Welch correction). The nitrate content of the vermicomposted WPS was very significantly higher than the windrow composted WPS at weeks 8 and 12 ($p < 0.001$, Student t-test with Welch correction). The vermicomposted WPS contained a 19.8-fold higher concentration of nitrate than the windrow composted WPS.

This resulted in final NH_4/NO_3 ratios of 0.021 ± 0.004 and 0.058 ± 0.004 , for windrow composted and vermicomposted WPS, respectively. The lower ratio for the windrow composted WPS reflected its proportionally lower NH_4^+ content, *cf.* 6 mg kg^{-1} for windrow composting with 357 mg kg^{-1} for vermicomposted WPS.

Changes in other macro-nutrients (P, K, Mg)

Table 42 – Phosphorous (P), potassium (K) and magnesium (Mg) contents

Composting method	Sampling period (week)	Total nutrients		Water extractable nutrients			Total nutrient ratios			
		P %	K %	P mg kg ⁻¹	K mg kg ⁻¹	Mg mg kg ⁻¹	C/P	C/K	N/P	N/K
Fresh	0	0.54±0.02	0.33±0.01	296±33	1277±40	360±11	83±1	134±4	5.0±0.2	8.1±0.4
Windrow	2	0.64±0.01	0.41±0.01	567±65	1693±75	96±12	61±2	94±2	2.7±0.1	4.2±0.2
	4	0.75±0.02	0.46±0.01	489±38	1876±30	77±3	47±2	77±2	2.4±0.1	3.9±0.2
	6	0.73±0.03	0.45±0.02	309±22	1741±48	111±5	47±2	76±4	2.4±0.2	4.0±0.3
	8	0.83±0.01	0.52±0.01	251±2	1531±85	63±8	37±1	59±2	2.3±0.1	3.6±0.1
	12	0.81±0.02	0.45±0.01	174±5	1480±33	71±14	36±1	65±2	2.6±0.1	4.8±0.2
VCU	2	0.78±0.03	0.46±0.02	727±88	2145±196	397±29	54±2	91±4	4.6±0.3	7.8±0.1
	4	0.95±0.05	0.69±0.08	845±46	2380±51	156±12	42±3	58±7	3.6±0.3	4.9±0.4
	6	0.95±0.05	0.49±0.03	745±14	2646±137	266±11	41±2	79±6	2.7±0.4	5.2±0.2
	8	0.95±0.02	0.42±0.02	639±41	2314±124	477±34	39±2	87±7	2.9±0.2	6.6±0.1
	12	0.90±0.03	0.39±0.01	445±32	2065±71	731±8	38±2	87±3	2.8±0.1	6.3±0.1

At all sampling periods windrow-composted WPS contained significantly ($p<0.05$, Student t-test) lower levels of total phosphorus (P %DM) than vermicomposted WPS (Table 42). Final total P losses (corrected for loss of dry mass) after 12 weeks amounted to $39.8\pm0.8\%$ ash and $17.3\pm0.4\%$ ash for windrow-composted and vermicomposted WPS, respectively; a 2.3-fold difference ($p<0.0001$, Student t-test).

Higher total P losses were concurrent with significantly lower levels of water extractable P ($p<0.001$, Student t-test) within the windrow composted WPS from weeks 4 to 12. Final water extractable P concentration in the vermicomposted WPS (445 mg kg^{-1}) was 2.6-fold higher ($p<0.001$, Student t-test) than windrow composted WPS (174 mg kg^{-1}). Vermicomposted WPS contained a 50% higher concentration of water extractable P than fresh WPS ($p<0.01$, Student t-test), whereas windrow composted WPS contained a 41% lower water extractable P concentration than fresh WPS ($p<0.05$, Student t-test).

Final C:P and N:P ratios, for windrow composted (36:1 and 2.6:1, respectively) and vermicomposted WPS (38:1 and 2.8:1, respectively), were significantly lower than initial values ($p<0.001$, Student t-test). This demonstrated carbon and nitrogen losses exceeded phosphorus losses during windrow composting (2.3 and 1.9-fold, respectively) and vermicomposting (2.2 and 1.8-fold, respectively). Despite a >2-fold greater loss in total P within the windrow composted WPS compared to vermicomposted WPS, final C/P and N/P ratios did not differ significantly ($p>0.05$, Student t-test). This indicated that greater losses in total P within windrow composted WPS were concurrent with proportionally greater losses of carbon and nitrogen losses (see previous sections).

At weeks 8 and 12, total potassium (K %DM) levels were significantly ($p<0.01$, Student t-test) lower for the vermicomposted WPS (0.42 and 0.39%) than windrow composted WPS (0.52 and 0.45%). However, when corrected for loss of dry mass (%ash), the windrow composted WPS showed significantly greater total K losses at all sampling periods ($p<0.01$, Student t-

tests). Final total K losses (corrected for loss of dry mass) after 12 weeks amounted to $46.6 \pm 0.9\%$ and $41.8 \pm 0.8\%$ for the windrow composted and vermicomposted WPS, respectively ($p < 0.01$, Student t-test).

Higher total K losses were concurrent with significantly lower levels of water extractable K ($p < 0.001$, Student t-test) within the windrow composted WPS at all sampling periods. Final water extractable K concentration in the vermicomposted WPS (2065 mg kg^{-1}) was 1.4-fold higher ($p < 0.001$, Student t-test) than windrow composted WPS (1480 mg kg^{-1}). Vermicomposted WPS contained a 62% higher concentration of water extractable K than fresh WPS ($p < 0.001$, Student t-test), whereas windrow composted WPS contained only a 16% higher water extractable K concentration than fresh WPS ($p < 0.05$, Student t-test).

Final C:K and N:K ratios, for windrow composted (65:1 and 4.8:1, respectively) and vermicomposted WPS (87:1 and 6.3:1, respectively), were significantly lower than initial values ($p < 0.01$, Student t-test). This demonstrated carbon and nitrogen losses exceeded phosphorus losses during windrow composting and vermicomposting.

Despite only a 11.5% greater loss in total K within the windrow composted WPS, final C/K and N/K ratios differed significantly ($p > 0.01$, Student t-test) between the windrow composted and vermicomposted WPS. This indicated that greater losses in total K within windrow composted WPS were not proportional to greater losses of carbon and nitrogen losses.

Levels of water extractable magnesium were significantly lower within the windrow composted WPS than vermicomposted WPS at all sampling periods ($p < 0.001$, Student t-tests). The final concentration of water extractable Mg in vermicomposted WPS (731 mg kg^{-1}) was 103% higher ($p < 0.001$, Student t-test) than for fresh WPS (360 mg kg^{-1}). Whereas, the final concentration of water extractable Mg in windrow-composted WPS (71 mg kg^{-1}) was 80% lower ($p < 0.001$, Student t-test) than fresh WPS.

Changes in other ionic factors - acidity (pH) and electrical conductivity (EC)

Table 43. – Changes in pH &EC during active composting

Composting method	Sampling period (weeks)	PH	[H ⁺] × 10 ⁻⁷	EC μS cm ⁻¹	Total salts (NO ₃ ;NH ₄ ;P;K;Mg) mg l ⁻¹
Fresh	0	7.2±0.1	0.73	1350±66	347±30
Windrow	2	7.8±0.2	0.21	1405±78	1328±80
	4	7.5±0.0	0.30	1403±52	1386±37
	6	6.8±0.0	1.50	585±6	677±21
	8	7.4±0.2	0.50	430±14	518±25
	12	6.9±0.0	1.19	430±9	497±19
VCU	2	6.0±0.0	11.5	1300±87	863±59
	4	7.3±0.1	0.59	1252±104	1244±113
	6	6.8±0.0	1.69	650±23	638±28
	8	6.5±0.2	3.82	970±68	1058±83
	12	6.2±0.0	6.72	1913±145	2218±145

Changes in acidity (pH)

The pH of the windrow composting WPS rose significantly from pH 7.2 to 7.8 by week 2, a 71% reduction in [H⁺] (p<0.05, Tukey-Kramer test). Subsequently, although pH levels fluctuated within the WPS during windrow composting, a general decrease was observed, reaching a final pH of 6.9 after 12 weeks. This final figure was significantly below an initial pH of 7.2 for fresh WPS, a 63% increase in [H⁺] (p<0.05, Tukey-Kramer test).

During vermicomposting the pH level of WPS fell significantly to pH 6.0 by week 2, a 15.8-fold increase in [H⁺] (p<0.001, Tukey-Kramer test). The pH increased by week 4 to pH 7.3 (p<0.001, Tukey-Kramer test). This was possibly related to concurrent increases in ammonium. The final pH of the vermicomposted WPS was 6.2, a 9.2-fold higher concentration [H⁺] than fresh WPS (p<0.001, Tukey-Kramer test).

The final pH of vermicomposted WPS (pH 6.2) was significantly below that for windrow composted WPS (pH 6.9), equating to a 5.6-fold higher concentration of [H⁺] (p<0.001, Tukey-Kramer test).

Changes in electrical conductivity (EC)

The EC within windrow composting WPS did not change significantly from fresh WPS (1350 μS cm⁻¹) until week 6, where it fell to 585 μS cm⁻¹ (p<0.001, Tukey-Kramer test). No further significant changes were observed, although a lower final EC of 430 μS cm⁻¹ was achieved.

The EC within vermicomposting WPS also did not change significantly from fresh WPS (1350 μS cm⁻¹) until week 6, where it fell to 650 μS cm⁻¹ (p<0.001, Tukey-Kramer test). At week 8, EC had increased significantly from week 6 to 970 μS cm⁻¹ (p<0.001, Tukey-Kramer test), which increased further by week 12 to 1913 μS cm⁻¹ (p<0.001, Tukey-Kramer test). These increases were concurrent with large levels of nitrate production.

The final EC value for vermicomposted WPS ($1913 \mu\text{S cm}^{-1}$) was 4.4-fold that of windrow composted WPS ($430 \mu\text{S cm}^{-1}$), and 42% higher than fresh WPS ($1350 \mu\text{S cm}^{-1}$).

7.2.2 Comparisons of the composted products.

Physicochemical composition

Table 44 - Physicochemical analyses of experimental plant growth media (determinations were obtained from single composite samples).

Plant growth media	Total nutrients ¹					Water-extractable nutrients ²						
						Macronutrients						
	C	N	C:N	P	K	NH ₄ ⁺	NO ₃ ⁻	P	K	S	Ca	Mg
Windrow compost	29	2.25	12.9	0.85	0.52	<1	132	51	392	156	174	27
50% coir mix	34	1.76	19.3	0.68	0.53	<1	6	54	413	91	92	22
Vermicompost	34	2.60	13.1	0.90	0.49	30	1500	111	452	222	1434	191
50%coir mix	37	2.21	16.7	0.78	0.51	12	840	84	472	102	714	122
100% Coir (control)	50	0.65	75.4	0.03	0.74	<1	3	5	143	96	12	4

Plant growth Media	Water-extractable ²			Water extractable nutrient ratios ³			Ionic factors		Physical factors		
	Micronutrients										
	Fe	Na	Cl	N:P	N:K	K:Mg	pH	EC ⁴	BD ⁵	AFP ⁶	SWC ⁷
Windrow compost	7	155	213	2.6	0.3	14.5	7.1	560	563	28	64
50%coir mix	8	203	298	0.1	0.01	18.8	7.2	520	507	19	75
Vermicompost	7	203	170	13.8	3.4	2.4	6.1	2100	615	27	69
50%coir mix	7	248	255	10.1	1.8	3.9	6.3	1410	561	20	75
100% Coir (control)	7	97	255	0.6	0.02	34	6.1	180	440	22	90

- 1. All figures = % dry mass;
- 2. All figures = mg l⁻¹ wet mass;
- 3. N:P and N:K ratios for extractable nutrients, N was taken as the sum of NH₄ and NO₃ contents;
- 4. Electrical-conductivity (µS cm⁻¹) at 25°C;
- 5. Bulk density g l⁻¹ wet mass;
- 6. Air-filled porosity (% by volume);
- 7. Saturated water content (% by volume).

Windrow compost contained lower levels of total carbon (29%DM) and total nitrogen (2.3%DM) compared to vermicompost (34%DM total carbon and 2.6%DM total nitrogen). This equated to a 15.6% difference in nitrogen and a 17.2% difference in carbon, resulting in broadly similar C:N ratios, 12.9 (windrow compost) and 13.1 (vermicompost). Total phosphorus (P) and potassium (K) contents were also broadly similar between composts (0.85 v. 0.90%DM and 0.49 v. 0.52%DM, respectively).

Differences in total macronutrients resulted in higher N:P and N:K ratios (in terms of %DM) for the vermicompost (2.9 and 5.3, respectively) compared to the windrow compost (2.6 and 4.3, respectively).

The most dramatic differences between the windrow compost and vermicompost were observed for water extractable nutrients and their effect on other ionic factors.

The vermicompost contained higher concentrations of all the water extractable macronutrients measured. These shall be expressed in terms of mg l^{-1} wet mass (WM), as this is considered more appropriate when assessing the levels of nutrients available to plant roots during cultivation (*q.v.* Bunt 1976).

Vermicompost contained 1500 mg l^{-1} WM water extractable nitrate (NO_3^-), 11.4-fold that of the windrow compost (132 mg l^{-1} WM). Vermicompost also contained water extractable nitrogen in the form of 30 mg l^{-1} WM ammonium (NH_4^+), compared to $<1 \text{ mg l}^{-1}$ WM in the windrow compost. This amounted to total water extractable nitrogen contents ($[\text{NH}_4^+] + [\text{NO}_3^-]$) of 1530 and 132 mg l^{-1} WM (0.66 and 0.05%DM) for the vermicompost and windrow compost, respectively.

The vermicompost contained 1434 mg l^{-1} WM water extractable calcium (Ca), 8.2-fold that of the windrow compost (174 mg l^{-1} WM). Vermicompost also contained 7.1-fold magnesium (Mg) and 2.2-fold phosphorus (P) compared to the windrow compost (in terms of mg l^{-1} WM). Water extractable sulphur (S – sulphate $[\text{SO}_4^{2-}]$) and potassium (K) were 42% and 15% higher in the vermicompost (in terms of mg l^{-1} WM).

Ratios of water extractable macronutrients (mg l^{-1} WM) reflected these differences, with high N:P and N:K ratios observed for the vermicompost (13.8:1 and 3.4:1, respectively) compared to the windrow compost (2.6:1 and 0.3:1, respectively). The windrow compost possessed a much higher K:Mg ratio (14.5:1) compared to the vermicompost (2.4:1), due to its proportionally lower Mg content.

Differences in water extractable micronutrients were less marked. Levels of water extractable iron (Fe) were broadly similar between windrow compost and vermicompost (7 v. 7 mg l^{-1} WM, respectively). The windrow compost contained 25% more water extractable chloride (Cl). Vermicompost contained a 55% higher concentration of sodium (Na) than windrow compost (203 v. 155 mg l^{-1} WM, respectively).

The total water extractable nutrient (salt) concentrations of the vermicomposted and windrow composted were 4320 and 1370 mg l^{-1} WM (1.86% and 0.48%DM) respectively; a 3.2-fold difference. As expected a broadly similar, 3.8-fold, difference in electrical conductivity (EC) was observed between the vermicompost ($2100 \mu\text{S cm}^{-1}$) and windrow compost ($560 \mu\text{S cm}^{-1}$).

The vermicompost was slightly acidic (pH 6.1), whereas the windrow compost was neutral (pH 7.1). These differences in pH equated to a 10-fold greater concentration of hydrogen ions in the vermicompost.

The physical characteristics of the windrow compost and vermicompost were broadly similar. Bulk densities were 563 and 615 g l⁻¹ WM for windrow compost and vermicompost, respectively; with an air-filled-porosity of 28 and 27%; and saturated-water-content of 64 and 69%, respectively.

The particle sizes of the windrow compost were 4% 5-10 mm, 72% 2-5 mm and 24% <2 mm. The vermicompost contained particle sizes of 95% 2-5 mm and 5% <2 mm.

Changes in mineral nutrient contents, upon the addition of coir, were observed for both windrow and vermicomposts.

Total water extractable nitrogen ($[\text{NH}_4^+] + [\text{NO}_3^-]$) and calcium concentrations appeared to be the most effected with decreases of 44% and 50% for the vermicompost, and 95% and 47% for windrow compost. The larger reduction in nitrogen shown during the dilution of windrow compost was attributed to the possible immobilisation of nitrogen by microbes, utilising carbon from in the coir, during compost preparation and analysis (Chapter 6).

Water extractable magnesium in the vermicompost/coir mix was 36% lower than the unmixed vermicompost; and levels were reduced by 19% in the windrow compost/coir mix.

Water extractable phosphorus and potassium were broadly unchanged in the windrow compost/coir mix (rose by 6 and 5%, respectively) compared to the unmixed windrow compost. However, phosphorus was reduced by 24% in the vermicompost/coir mix compared to unmixed vermicompost; whereas, potassium remained relatively unchanged (rose by 4%).

Both water extractable sodium and chloride was increased in the windrow compost and vermicompost derived medium by the addition of coir. Water extractable sodium was increased by 31% and 22% by mixing with coir for windrow compost and vermicompost, respectively; and water extractable chloride increased by 40% and 50%, respectively.

The total dilution of water extractable nutrients (salts) by mixing vermicompost with coir, amounted to a 1464 mg l⁻¹ (34%) decrease, consistent with a 690 $\mu\text{S cm}^{-1}$ (33%) decrease in EC.

The total dilution of water extractable nutrients (salts) by mixing windrow compost with coir, amounted to a 138 mg l⁻¹ (10%) decrease, broadly consistent with a 40 $\mu\text{S cm}^{-1}$ (7%) decrease in EC.

The effect of coir addition upon the physical properties of the windrow compost and vermicompost derived growth media were very similar. Bulk density was decreased by 11% and 10%. Air-filled-porosity (AFP) was decreased to 19% and 26%; and saturated-water-contents were increased to 75%.

Plant growth responses

Figure 42 – Germination (7 days)

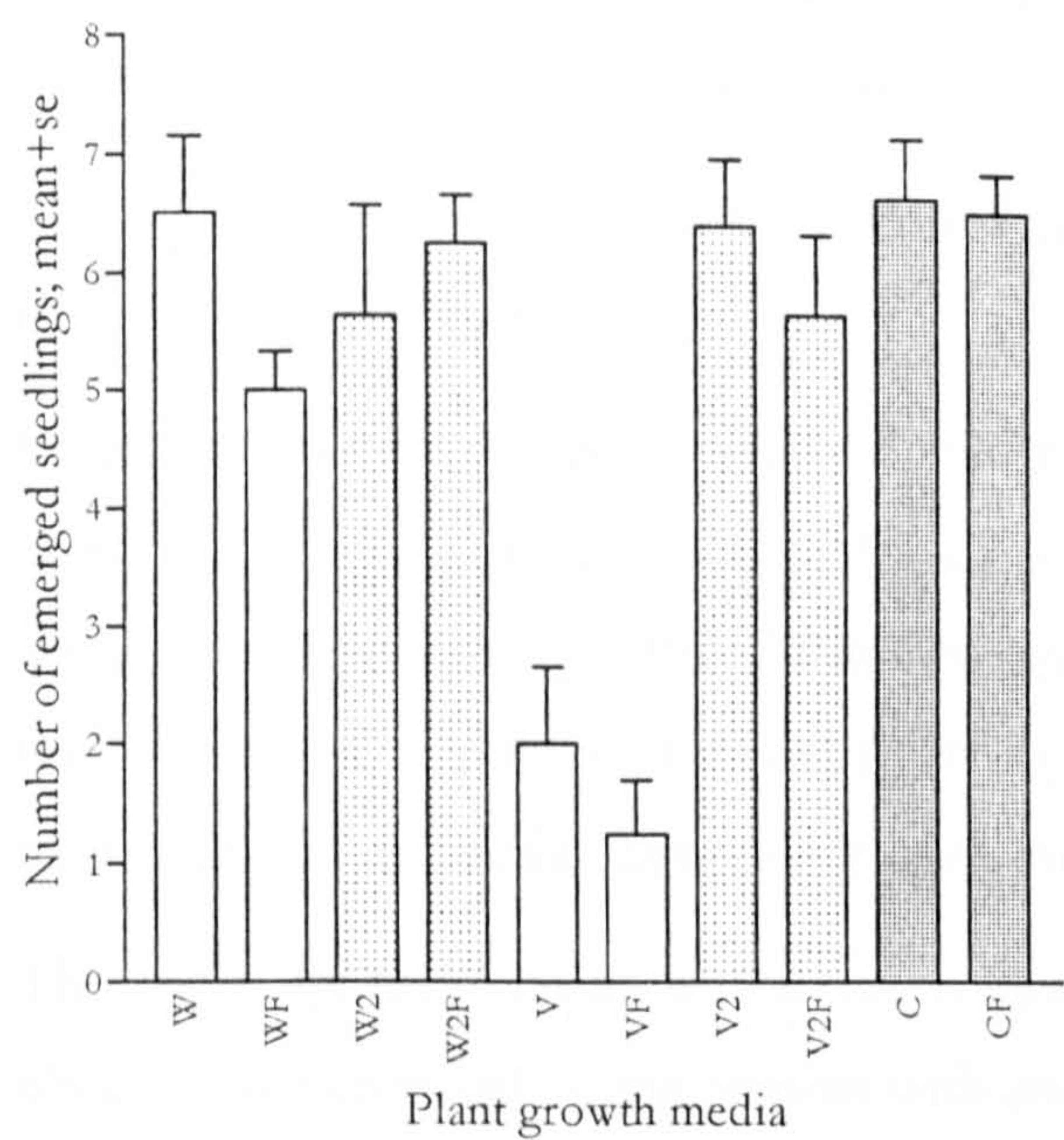


Figure 43 – Germination (14 days)

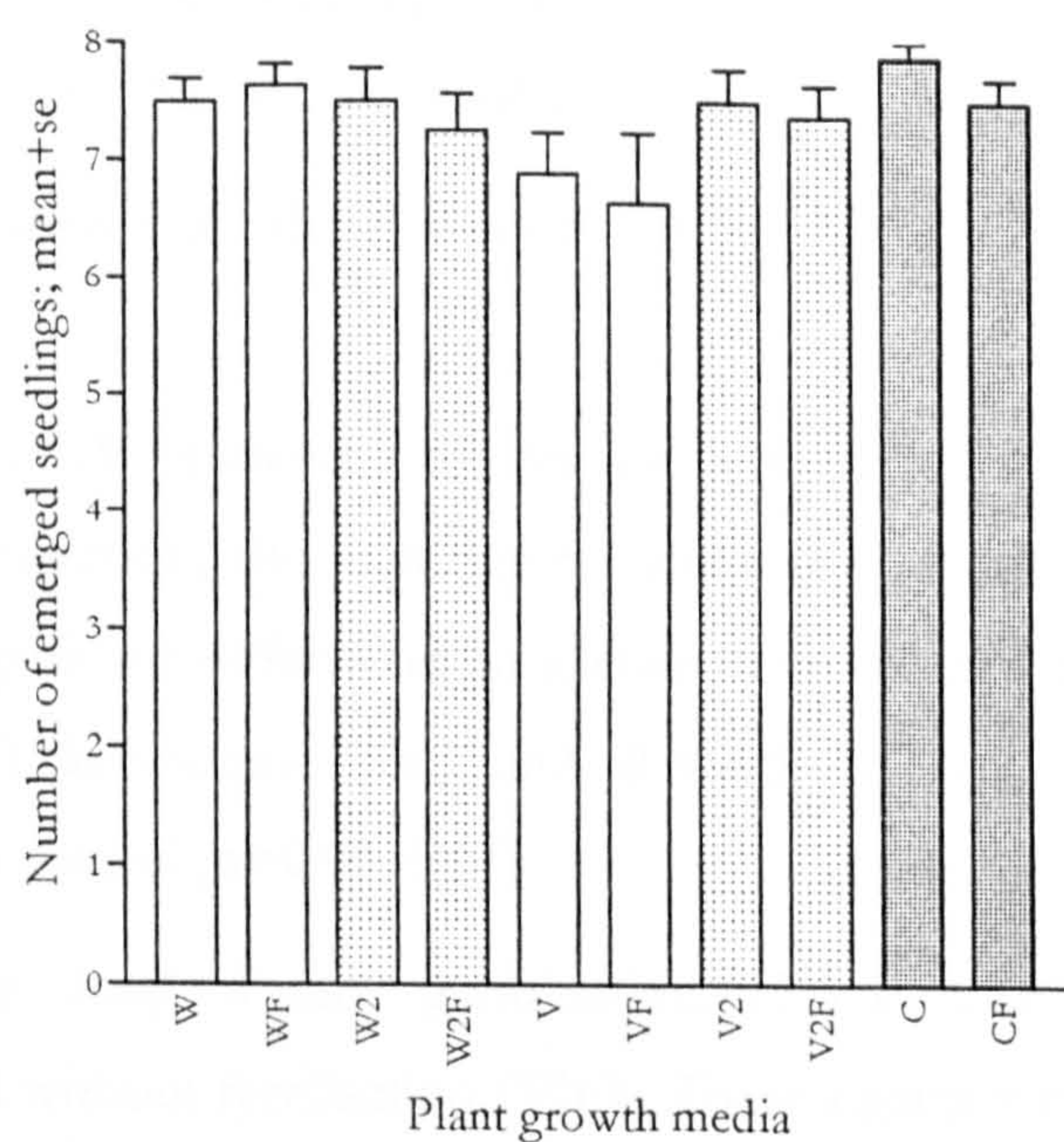


Table 45 - Radish germination and cotyledon diameter (mean±se).

Treatments		Germination ¹ (%)		Cotyledon diameter (mm)
Plant growth medium	Fertiliser added (F)	7 days (early)	14 days (total)	14 days
Windrow compost (W)	-	81.3±8.2	93.8±2.4	16.7±0.6
	F	62.5±4.1	95.3±2.3	15.9±0.5
50% coir mix (W2)	-	70.3±9.8	93.8±3.3	18.1±0.8
	F	78.1±5.2	90.6±3.9	17.7±0.5
Vermicompost (V)	-	25.0±8.2	85.9±4.4	13.0±0.5
	F	15.6±5.7	82.8±7.5	12.2±0.4
50% coir mix (V2)	-	79.7±7.1	93.8±3.3	15.7±0.4
	F	70.3±8.5	92.2±3.3	14.5±0.4
Coir-control (C)	-	82.8±6.2	98.4±1.6	15.2±0.4
	F	81.3±4.1	93.8±2.4	16.4±0.5

Early germination (7 days after sowing)

Results of early germination, after 7 days (Figure 42 and Table 45), showed there were significant variations between treatments (one-way ANOVA: $p < 0.0001$, $F = 11.3$, $R^2 = 59.2\%$).

The highest level of early germination was observed for the coir-control (C[F]) with 81.3 to 82.8% germination after 7 days.

Unmixed windrow compost without fertilization (W) showed a comparable level germination at 81.3%. This was decreased to 62.5% by fertilization, although this reduction was not quite statistically significant ($p = 0.07$, STW). No significant differences in germination affected by the addition of coir were observed ($p > 0.05$, Tukey-Kramer test), and all windrow compost treatments did not differ significantly from the control ($p > 0.05$, DM).

The most apparent result was a much lower level of early germination, 15.6 to 25.0%, observed for unmixed vermicompost with and without fertilization (V[F]). These figures were significantly below all other treatments ($p < 0.05$, Tukey-Kramer test). Amendment with coir significantly increased early germination in the vermicompost to 70.3 to 79.7% ($p < 0.01$, Tukey-Kramer test); which was not significantly different from the coir-control ($p > 0.05$, DM).

Total germination (14 days after sowing)

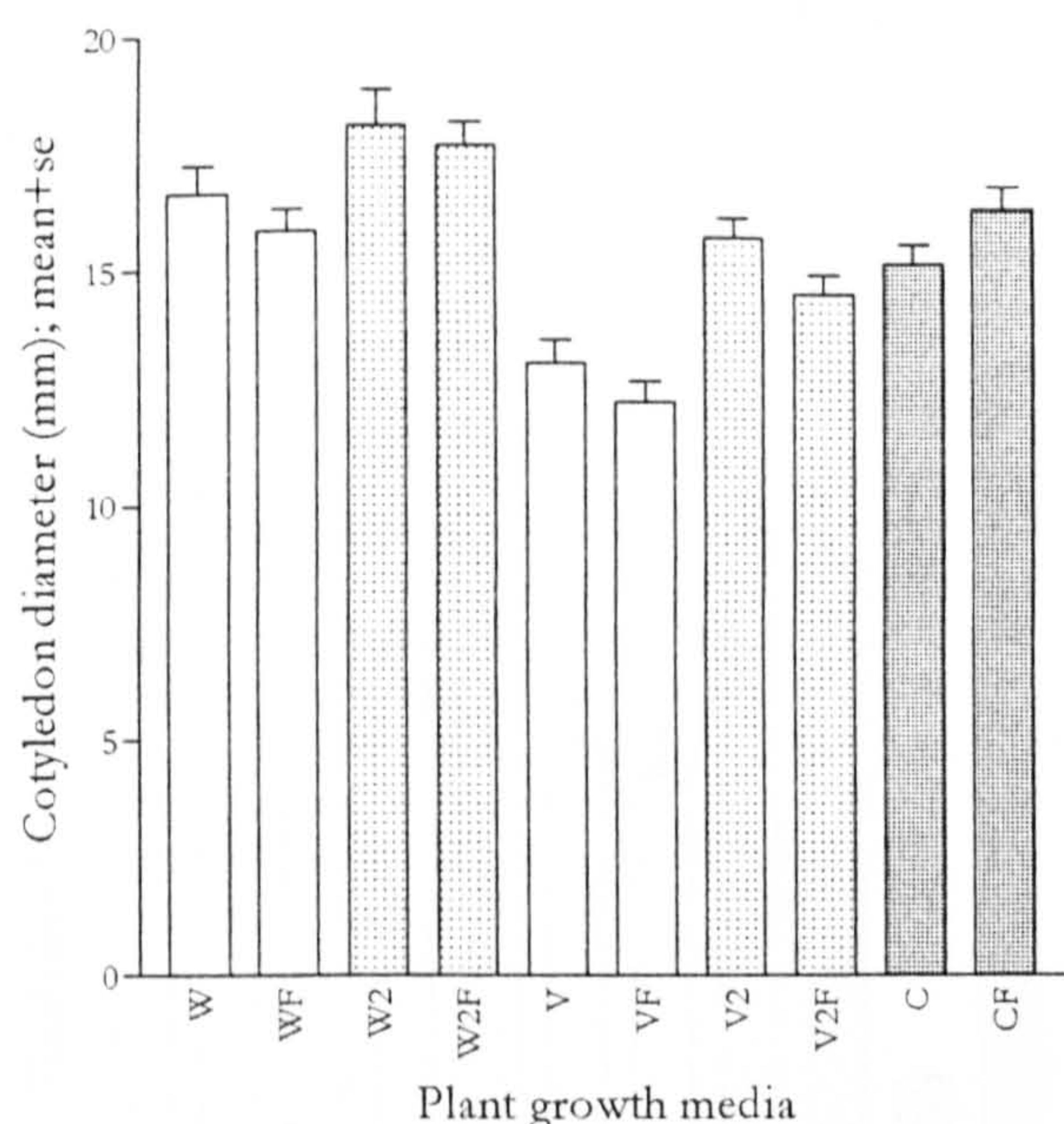
Total germination was assumed to have occurred after 14 days.

Variations in germination between treatments were reduced, and were not significant (one-way ANOVA: $p > 0.05$). This showed that differences in early germination (7 days) between treatments was due to delayed germination rather than its permanent inhibition.

A comparison of early and total germination values revealed significant differences for unmixed windrow compost with fertilization (WF) and unmixed vermicompost with and without fertilization (V[F]), only ($p < 0.05$, Tukey-Kramer test). This revealed that these treatments resulted in significantly delayed germination.

It was also noted that, although not statistically significant ($p > 0.05$, Tukey-Kramer test), unmixed vermicompost treatments (V[F]) resulted in mean total germination of 82.8 to 85.9%, whereas all other treatments resulted in total germination of $> 90\%$. This was supported by a two-way ANOVA, which showed that germination in vermicompost was significantly increased by coir addition, to 92.2 to 93.8% (*cf.* V[F] with V2[F], $p < 0.05$); no significant effect was attributed to fertilization or an interaction between treatments ($p > 0.05$). This showed that a small amount of permanent germination inhibition had occurred in the unmixed vermicompost (8 to 9%).

Figure 44 - Cotyledon diameter (14 days)



Early levels of plant growth, cotyledon diameter after 14 days (Figure 44 and Table 45), varied significantly between treatments (one-way ANOVA: $p < 0.0001$, $F = 11.3$, $R^2 = 59.2\%$).

Windrow compost treatments showed the highest cotyledon diameters, ranging from 15.9 to 18.1 mm, although, these figures were not significantly different from the fertilised coir-control (C[F]) ($p > 0.05$, Tukey-Kramer test). All windrow compost treatments resulted in significantly higher cotyledon diameters than all unmixed vermicompost treatments ($p < 0.01$, Tukey-Kramer test). Coir amended windrow-compost resulted in larger cotyledon diameters (17.7 to 18.1 mm) than coir amended vermicompost (14.5 to 15.7 mm) when under similar fertilization treatments ($p < 0.05$, Tukey-Kramer test). Cotyledon diameters in windrow compost were significantly increased by coir-amendment (two-way ANOVA: *cf.* W[F] with W2[F]), $p < 0.001$); whereas fertilization or an interaction between treatments had no significant effect ($p > 0.05$).

Unmixed vermicompost produced seedlings with mean cotyledon diameters (12.2 to 13.0 mm) which were significantly below all other treatments ($p < 0.01$, Tukey-Kramer test), except the unfertilised coir-control (C) and the fertilised coir-amended vermicompost (V2F) ($p > 0.05$, Tukey-Kramer test). Cotyledon diameters in vermicompost were significantly increased by coir-amendment (two-way ANOVA: *cf.* V[F] with V2[F]), $p < 0.0001$); whereas fertilization reduced cotyledon diameter ($p < 0.05$) with no significant interaction between treatments ($p > 0.05$).

Differences in cotyledon diameter between vermicompost treatments were mainly attributed to variations in early germination rates (Pearson’s correlation: $p<0.05$, $r=0.97$, $R^2=95\%$, $N=4$); whereas, early germination did not explain variations between windrow compost treatments (Pearson’s correlation: $p>0.05$).

Figure 45 – Total plant growth

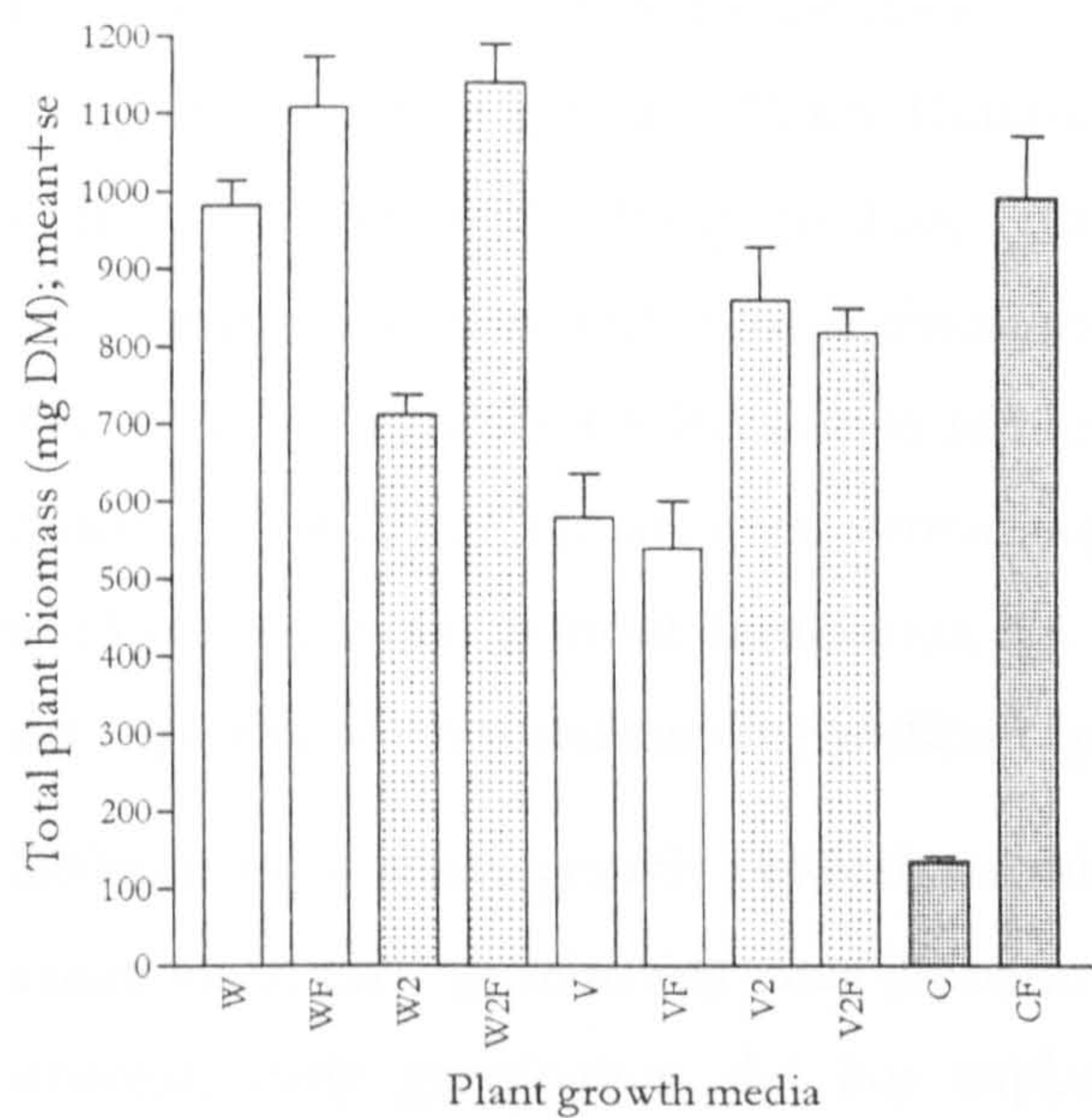


Figure 46 – Biomass partitioning (S:H ratio)

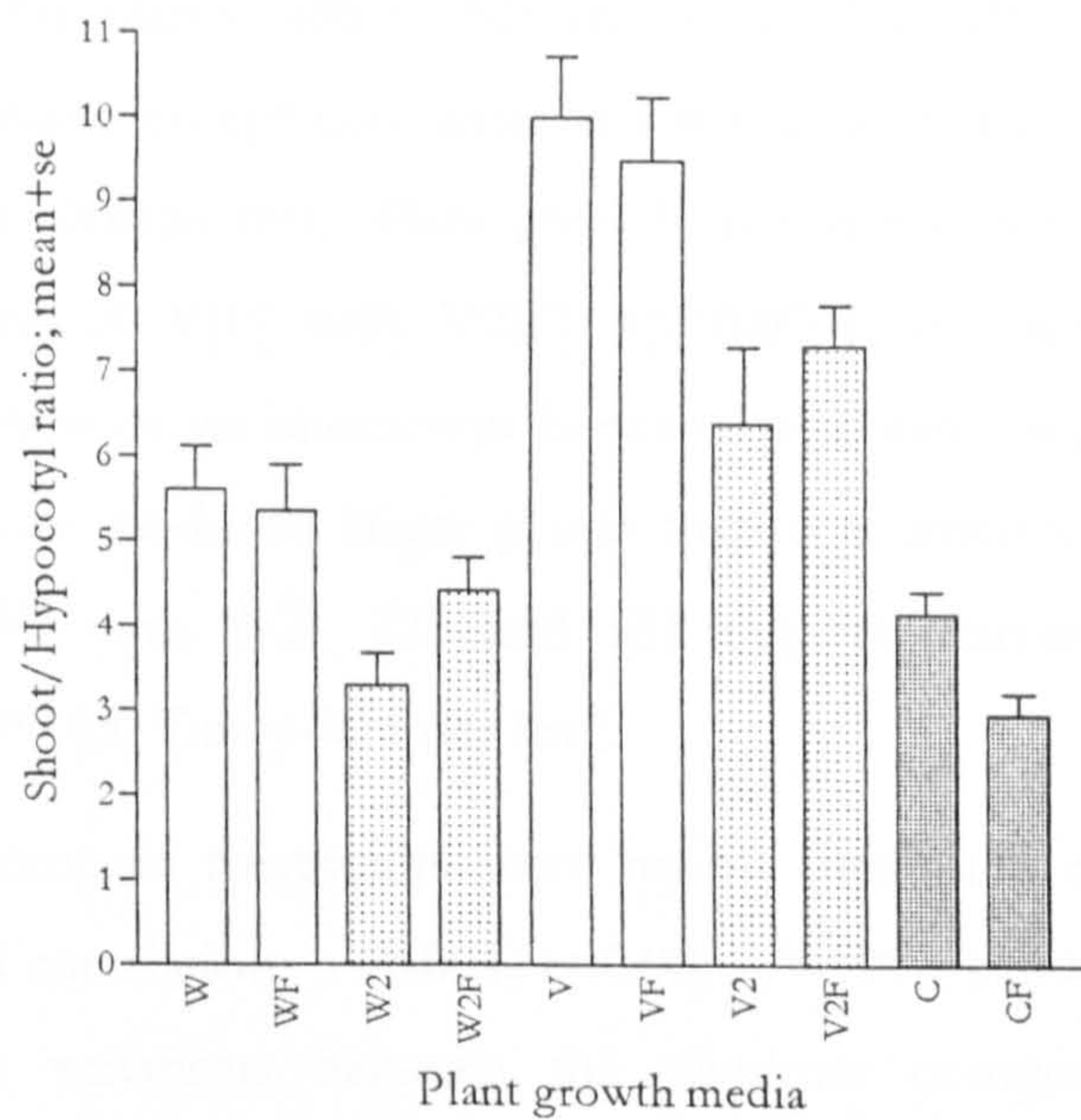


Table 46 - Total plant growth and biomass partitioning (mean±se)

Treatments		Individual plant dry mass (mg) after 21 days			
Plant growth medium	Fertiliser added (F)	Whole plant	Biomass partitioning		
			Shoot (S)	Hypocotyl (H)	S:H ratio
Windrow compost	–	981±33	823±23	158±18	5.6±0.5
	F	1108±65	919±41	189±26	5.4±0.5
50% coir mix	–	710±27	531±21	180±24	3.3±0.4
	F	1140±48	920±33	220±22	4.4±0.4
Vermicompost	–	579±57	525±52	55±7	10.0±0.7
	F	540±60	486±53	55±9	9.5±0.8
50% coir mix	–	860±67	725±48	135±24	6.4±0.9
	F	819±31	717±27	102±9	7.3±0.5
C	–	134±7	108±5	27±3	4.2±0.3
	F	992±81	726±49	267±41	3.0±0.3

Total plant growth (plant biomass after 21 days)

Significant variation in total plant growth (Figure 45 and Table 46) was observed between treatments (one-way ANOVA: $p<0.0001$, $F=34.4$, $R^2=81.6\%$).

The mean biomass of plants cultivated in amended and unmixed windrow compost, with fertilization (WF and W2F), 1108 to 1140 mg, were significantly higher than plants grown in all vermicompost treatments (V[F] and V2[F]), 540 to 860 mg ($p<0.05$, Tukey-Kramer test); but not the fertilised coir-control, 992 mg ($p>0.05$, Tukey-Kramer test).

Without fertilization, windrow-derived media produced smaller plants, 710 to 981 mg, *cf.* W with WF, and W2 with W2F ($p < 0.0001$, two-way ANOVA). A significant interaction between treatments ($p < 0.01$, two-way ANOVA) related to significantly reduced plant growth in coir-amended windrow compost without fertilization, *cf.* W with W2 ($p < 0.05$, Tukey-Kramer test), which was not observed with fertilization, *cf.* WF with W2F ($p > 0.05$, Tukey-Kramer test).

Unmixed vermicompost (V[F]) produced smaller plants, 486 to 525 mg, than all windrow-compost treatments ($p < 0.05$, Tukey-Kramer test), except coir amended windrow compost without fertiliser (W2), 710mg ($p > 0.05$, Tukey-Kramer test). Plant growth in vermicompost was significantly increased by coir-amendment, *cf.* V[F] with V2[F]) ($p < 0.0001$, two-way ANOVA); no significant effect due to fertilization or an interaction between treatments was observed ($p > 0.05$). Coir-amended vermicompost produced larger plants than coir-amended windrow compost, without fertilization (*cf.* V2 with W2), 725 and 531 mg respectively, although this was not statistically significant ($p > 0.05$, Tukey-Kramer test).

Differences in plant growth between vermicompost treatments were mainly attributed to variations in early germination rates (Pearson's correlation: $p < 0.001$, $r = 0.99$, $R^2 = 99\%$, $N = 4$); whereas, early germination did not explain variations between the windrow compost treatments (Pearson's correlation: $p > 0.05$).

Biomass partitioning (Shoot:hypocotyl)

The shoot:hypocotyl ratio (Figure 46 and Table 46), showed the ratio of the total biomass partitioned to shoot (leaves) and hypocotyl (bulb) during the 21 day experimental period. Significant variation between treatments was observed ($p < 0.0001$, $F = 7.5$, $R^2 = 50.0\%$; one-way ANOVA).

All windrow compost treatments (W[F] and W2[F]) produced plants with significantly lower S:H ratios (3.3 to 5.6) than all unmixed vermicompost treatments, 9.5 to 10.0 ($p < 0.001$, Tukey-Kramer test). Coir-amended windrow compost (W2[F]) resulted significantly ($p < 0.05$, Tukey-Kramer test) lower S:H ratios (5.4 to 5.6) than coir-amended vermicompost (V2[F]), under similar fertilization treatments (7.3 to 6.4, respectively). All windrow compost treatments did not differ significantly ($p > 0.05$, DM) from the coir-controls (C[F]). The S:H ratio of plants cultivated in windrow compost was significantly reduced by coir-amendment, *cf.* W[F] with W2[F] ($p < 0.01$, two-way ANOVA); no significant effect due to fertilization or an interaction between treatments was observed ($p > 0.05$).

Higher S:H ratios observed for plants cultivated in vermicompost were also significantly above the coir-controls ($p < 0.05$, DM), except for unfertilised vermicompost mixed with coir (V2). The S:H ratio of plants cultivated in vermicompost was significantly reduced by coir-

amendment, of V[F] with V2[F]) ($p < 0.001$, two-way ANOVA); no significant effect due to fertilization or an interaction between treatments was observed ($p > 0.05$).

7.3 Discussion

7.3.1 Comparisons between windrow and vermicomposting processes.

Temperature and moisture content

The most prominent abiotic differences between windrow composting and vermicomposting was processing temperature and moisture content of WPS.

Temperatures of the WPS during windrow composting were in excess of 50°C for the first 4 weeks, followed by temperatures of between 36 and 45°C for the remainder of the 12 week processing period. Vermicomposting temperatures were controlled at approximately 21°C.

These temperature differences were consistent with rapid moisture losses in the windrow composted WPS during 8 weeks turning (78 to 55%); falling to 52% after maturation (week 12).

The moisture content of WPS during 8 weeks vermicomposting remained high (78 to 84%). This was not only consistent with lower process temperatures, but was also due to moisture retention by the plastic vermicomposting units. However, when the vermicomposted WPS was removed and matured (weeks 8 to 12) it dried rapidly to a 64% moisture content.

Total organic matter (TOM)/carbon losses

A significantly greater overall TOM loss was observed for windrow composted WPS after 8 weeks (70%ash). This was consistent with a 70%ash loss in total fibre content. The largest fibre losses were hemicellulose (44%) and cellulose (41%). Although lignin appeared to increase by 2.8%DM, correcting for dry matter losses (correcting for ash) revealed a loss of 42%ash.

Vermicomposting resulted in a significantly lower, 53%ash, loss of TOM after 8 weeks ($p < 0.001$), consistent with a 54%ash loss in total fibre. Largest fibre losses were observed in cellulose (34%), with hemicellulose decreasing by only 10%. Again lignin appeared to increase (2.3%DM), but when corrected for ash, revealed a loss of 22%ash.

All fibre losses were significantly greater during windrow composting ($p < 0.01$).

After maturation (12 weeks), windrow composted WPS showed a final loss in TOM of 74%ash, significantly higher than 62%ash for vermicomposted WPS ($p < 0.001$).

Final TOM contents were 52%DM (29%DM carbon) for windrow compost and 62%DM (34%DM carbon) for vermicompost.

Non-linear regression predicted that if TOM losses continued along current trends (shown during the 12 week experimental period), prolonged maturation (after 12 weeks) of windrow composted and vermicomposted WPS might result in broadly similar final TOM values. These final values were estimated as 47.1 and 49.9%DM, respectively.

For windrow composting the actual TOM value after 12 weeks was 5.3%DM above the final value predicted by non-linear regression. Whereas for vermicomposting the difference was greater - 11.8%DM. This suggested the windrow compost was nearer to a final TOM content, and was therefore more stabilised.

This was consistent with TOM losses (%ash) between week 8 and 12 (*i.e.*, during maturation). No statistically significant loss in TOM was observed during the maturation of windrow compost ($p > 0.05$), whereas, a significant loss in TOM was observed during the maturation of vermicompost ($p < 0.001$).

Non-linear regression curves also showed a greater rate of TOM loss (%DM) occurred during windrow composting, with a constant half-life of 4.6 weeks, compared to an 8.7 week half-life for TOM during vermicomposting ($p < 0.05$).

A lower total TOM reduction and rate of TOM reduction shown by vermicomposting was mainly attributed to its much lower processing temperature.

Jackson & Line (1997b) showed that the time to produce a mature compost (based on CO₂ respiration) from paper-sludge could be shortened by 30-50 days using an incubation temperature of 55°C, compared to an incubation temperature of 35°C.

Similar effects of temperature on decomposition rates have also been observed in earlier bench scale studies, using other wastes (*e.g.*, Suler & Finstein 1977).

It is known that many decomposer organisms can tolerate high temperatures, *e.g.* thermotolerant organisms, with optima of 45 to 55°C, often coinciding with greatest organic matter losses (Swift *et al.* 1979). A similar range of temperatures was observed during the windrow composting of WPS during this study.

Mesophilic fungi with temperature optima of 20 to 25°C (Dix & Webster 1995) and mesophilic bacteria with optima of 25 to 40°C (Pelczar *et al.* 1986) were most likely present during vermicomposting. However, these organisms could also have been present during windrow composting, in the cooler layers of the windrow, *e.g.* the outer surface and bottom of the pile, especially during the latter stages of windrow composting and maturation.

The wider range of temperatures experienced by the WPS during windrow composting may explain the larger losses of cellulose observed. The WPS exposed to a wider range of temperatures and therefore a greater mixture of organisms, producing a potentially

complementary mixture of extracellular enzymes. It has been demonstrated experimentally that a mixed culture of fungi increases the rate of cellulolysis (Dix & Webster 1995).

It is also likely that some of the organic matter loss during windrow composting may have been due to the leaching of water-soluble forms of carbon during the early stages of windrow composting. Frederickson (1997) found that the COD and BOD of green-waste compost leachate was highest during the early stages of composting, indicating high levels of organic and inorganic compounds. Diaz & Trezek (1979) found a similar phenomenon during mixed refuse/sludge composting.

The high water concentration of the WPS during vermicomposting may have decreased organic matter losses. The compaction of WPS was also observed during vermicomposting as the particle size of the material decreased, and the weight of the upper layers of WPS compacting lower layers. A combination of these two factors undoubtedly created areas (microclimates) of anaerobicity in the WPS during vermicomposting. The anaerobic decomposition of organic matter is much slower than aerobic decomposition (Mitchell 1974). This was consistent with a significant loss of organic matter when vermicomposted WPS was removed from vermicomposting units (VCUs) to mature and air-dry, resulting in the increased aeration of the material.

Nitrogen losses

Total nitrogen losses during 8 weeks active windrow composting and 4 week maturation period (68%ash) were also significantly greater ($p < 0.001$) than observed during vermicomposting (55%ash).

However, these losses were proportional to their respective carbon losses (74 and 62%ash), resulting in similar final C:N ratios of 13.5:1 and 13.7:1, respectively.

Greater nitrogen losses during windrow composting were consistent with higher process temperatures and pH levels during the early stages of composting (first 4 weeks). This led to ammonia production, consistent with the analysis of interstitial gases (Chapter 4). The loss of nitrogen through ammonia was consistent with large reductions in ammonium.

Nitrogen losses during windrow composting were typically high, *e.g.*, up to 70% (*q.v.* Lopez-Real 1990). High nitrogen losses resulted in compost with low water extractable nitrogen contents - 6 mg kg⁻¹ [NH₄⁺] and 308 mg kg⁻¹ [NO₃⁻].

Lower nitrogen losses during vermicomposting, consistent with a much lower processing temperature and lower pH values, resulted in compost with much higher water extractable nitrogen contents - 357 mg kg⁻¹ [NH₄⁺] and 6111 mg kg⁻¹ [NO₃⁻].

Low final $\text{NH}_4\text{:NO}_3$ ratios (after 12 weeks) were recorded for both windrow composted and vermicomposted WPS, 0.02:1 and 0.06:1 respectively. This suggested high levels of compost maturity had been obtained during both processes, despite lower TOM losses observed during vermicomposting.

Lower levels of nitrate in the windrow compost may also have been due to the inhibition of the nitrification process by higher maturation temperatures. Nitrification is regarded to be inhibited at temperatures $>40^\circ\text{C}$ (De Bertoldi *et al.* 1982b).

Changes in other macronutrients during composting

Windrow composting resulted in greater total phosphorus and potassium losses than vermicomposting.

Windrow composting resulted in a 40%ash loss of total phosphorus and a 47%ash loss of total potassium; whereas, vermicomposting resulted in a 17%ash loss of total phosphorus and a 42%ash loss of total potassium.

The lower mineral nutrient losses during vermicomposting were consistent with higher water extractable contents of mineral nutrients in the final vermicomposted WPS, after 12 weeks – 445 mg kg^{-1} phosphorus and 2065 mg kg^{-1} potassium.

The windrow composted WPS contained significantly lower levels of these nutrients after 12 weeks ($p<0.01$) - 174 mg kg^{-1} phosphorus and 1480 mg kg^{-1} potassium.

Water extractable magnesium was much lower in the windrow composted WPS after 12 weeks (71 mg kg^{-1}) compared to the vermicomposted WPS (731 mg kg^{-1}).

Losses of total phosphorus and potassium during windrow composting were likely to be due to nutrient leaching (*q.v.* Frederickson 1997), concurrent with leachate formation during the early stages of the windrow composting of WPS, observed during this study.

Lower losses observed during vermicomposting suggested less nutrient leaching occurred. However, potassium losses were only 10% lower than during windrow composting, whereas phosphorus loss was 57% lower. As these elements are taken up by earthworms at similar rates (*q.v.* Hartenstein & Hartenstein 1981), relative differences in their losses probably related to differences in the mobility of their water extractable forms. The solubility of phosphorus may have been reduced by its precipitation with high levels calcium (see discussion, Chapter 6).

Other ionic changes

High pII levels observed during the early stages of windrow composting (first 4 weeks), were consistent with the production of ammonia (see above). The pH of WPS during

vermicomposting was generally lower than during windrow composting. The lower pH of vermicomposting WPS may relate [not only to lower ammonia production] but also the formation of organic acids, intermediate substances produced during decomposition (*q.v.* Ndegwa *et al.* 2000). The formation of organic acids, such as volatile fatty acids (*e.g.* acetic acid), may have been exacerbated by pockets of anaerobicity (*q.v.* Domeizel *et al.* 1996), caused by the high moisture content and compaction of WPS during vermicomposting.

Organic acids formed during decomposition may have been leached during windrow composting and counteracted by the production of ammonia, a strong alkali. It is unlikely that losses of organic acids (*e.g.* volatile fatty acid) were lost via volatilisation, as these acids are usually present in the form of anions at high pH (6-9), thereby reducing evaporation (Lechner & Binner 1995).

The final pH level of the vermicompost was pH 6.2 after 12 weeks. This was significantly lower than a final pH of 6.9 observed for the windrow composted WPS after 12 weeks. The lower pH of the vermicomposted WPS was attributed to very high levels of nitrification.

Ndegwa *et al.* (2000) attributed pH reductions during the vermicomposting of biosolids to the production of nitrites/nitrates [$\text{NO}_2^-/\text{NO}_3^-$] and orthophosphates [H_2PO_4^- and HPO_4^{2-}].

This was consistent with the higher levels of water extractable nitrate and phosphorus of the vermicompost observed during this study.

A lowered pH in vermicomposted wastes has also been observed in other studies (Hartenstein & Hartenstein 1981; Mitchell 1997).

The high electrical conductivity (EC) of the vermicompost after 12 weeks ($2220 \mu\text{S cm}^{-1}$) was >4-fold higher than the final windrow compost ($500 \mu\text{S cm}^{-1}$). This reflected the lower total water extractable nutrients (salts) in the windrow compost compared to the vermicompost.

7.3.2 Comparisons of the composted products.

Physicochemical composition of windrow compost and vermicompost derived plant growth media

The physicochemical composition of the vermicompost and windrow compost used in radish plant cultivation trials was consistent with the composition of the final composted products after 12 weeks (see Chapter 6). However, reductions in ammonium and increases in nitrate suggested the continued progression of the nitrification process in both composts (*op. cit.*).

Final total water extractable nitrogen contents were 1530 mg l^{-1} for the vermicompost, and 132 mg l^{-1} for the windrow compost, a 11.6-fold difference. Magnesium was also much higher in the vermicompost at 191 mg kg^{-1} compared to 27 mg l^{-1} for the vermicompost, a 7-fold

difference. Water extractable phosphorus did not vary as considerably being 118% greater in the vermicompost (111 mg l⁻¹) than the windrow compost (51 mg l⁻¹). Water extractable potassium content was only 15% greater in the vermicompost (452 mg l⁻¹) compared to the windrow compost (392 mg l⁻¹).

Other mineral nutrients not monitored during the composting processes also differed greatly between the composts. The vermicompost contained an approximately 8.2-fold larger concentration of water extractable calcium (1434 mg l⁻¹) compared to the windrow compost (174 mg l⁻¹). The vermicompost also contained 42% high levels of water extractable sulphur (222 mg l⁻¹) than windrow compost (156 mg l⁻¹). The micronutrients sodium was also higher in the vermicompost by 55%; whereas iron and chloride contents were similar for both composts.

Large differences in levels of water extractable mineral nutrients between composts were reflected in very different ratios of mineral nutrient.

Low levels of water extractable nitrogen in proportion to other mineral nutrients in the windrow compost were consistent with low N:K (2.6:1) and N:P (0.3:1) ratios, compared to vermicompost: N:P (14:1) and N:K (3.4:1). Windrow compost also possessed a much higher K:Mg ratio (15:1) than the vermicompost K:Mg ratio (2.4:1).

High levels of nitrogen and its balance with other nutrients in the vermicompost was higher than levels recommended for plant mineral nutrition, as was its EC which reflected high levels of all mineral nutrients.

The windrow compost was much lower in water extractable nitrogen and magnesium, which were lower than recommended levels and correspondingly unbalanced with normal potassium concentrations, in terms of the general nutritional requirements of plants.

The physical characteristics of the composts, such as bulk-density, air-filled porosity, and saturated-water-content were similar. However, the windrow compost contained a wider range of particle sizes (>5 mm to <2mm) compared to the vermicompost (2-5 mm). The windrow compost also contained a larger proportion (24%) of fine material (<2 mm), compared to the vermicompost (5%).

The physicochemical composition of the windrow compost and vermicompost derived media were altered by blending with coir. The main changes experienced for both composted WPS derived media were decreases in water extractable mineral nutrients. Many nutrients were approximately halved, *i.e.*, water extractable nitrogen and calcium. Other nutrients were diluted by a lesser extent, *i.e.*, phosphorous, potassium, sulphur and magnesium. And some water extractable nutrient levels increased, *i.e.*, sodium and chloride.

Variation in the dilution of nutrients between composts, when mixed with coir, were probably due to large variation in the concentrations of nutrients present, and corresponding antagonistic effects upon CEC and water extractability (see Chapter 6).

The greater dilution of nutrients (mg l^{-1}), especially nitrogen and calcium, explained the greater effect of coir addition on the EC of vermicompost derived media (34% reduction). The mixing of windrow compost with coir had a lesser effect due to much lower initial concentrations of these nutrients.

Plant growth responses

The germination of radish seeds sown in the unmixed vermicompost was significantly delayed ($p < 0.05$); with only 16 to 25% germination after 7 days, compared to 70 to 80% in the vermicompost/coir mix (a 4-fold increase in germination). This was attributed to the high EC and osmotic effects of the vermicompost due to its high nutrient (salt) content, which was greatly reduced when mixed with coir.

Germination was not delayed in the windrow compost derived media, showing levels of germination broadly similar to that of the coir controls. However, fertilised unmixed windrow compost did show some delay in germination, increasing significantly from 63% after 7 days to 95% after 14 days ($p < 0.05$).

Total germination after 14 days was high for most media, resulting in %-germination of $>90\%$. The unmixed vermicompost showed the lowest total germination at 83 to 86%, which though not significantly different from windrow compost derived media, was significantly increased to 92 to 94% by the addition of coir ($p < 0.05$, two-way ANOVA). This suggested that some level of permanent inhibition of germination had occurred in the unmixed vermicompost.

Windrow compost derived media produced significantly higher levels of early radish growth (cotyledon diameters) than vermicompost under similar coir and fertiliser treatments ($p < 0.05$).

This showed that plant growth was reduced in the coir-mixed vermicompost despite showing similar rates of germination to the coir-mixed windrow compost media.

Total radish growth showed large differences between the windrow compost and vermicompost derived cultivation media.

Fertilised windrow compost derived media produced the largest radish plants (1108 to 1140 mg DM), compared to 710 to 981 mg without fertilization ($p < 0.0001$, two-way ANOVA). This was attributed to low nutrient levels and an imbalance of nutrients in the windrow compost, alleviated by fertilization, especially when the compost was mixed with coir.

The largest radishes produced in the vermicompost derived media were in the coir mix (819 to 860 mg DM), compared to 486 to 525 mg in the unmixed vermicompost ($p < 0.0001$, two-way ANOVA). This was attributed again to high levels of nutrients in the vermicompost, diluted to less harmful levels when the vermicompost was mixed with coir. Lower growth was also attributed to delayed germination (after 7 days) which was significantly correlated to total plant growth ($p < 0.001$).

Nutrient stresses upon plant growth was also revealed by plant biomass partitioning (shoot:hypocotyl - S:H ratios).

As expected the highest S:H ratios ('plant stress') was observed for unmixed vermicompost (9.5:1 to 10:1), significantly higher than the unmixed windrow compost (5.4:1 to 5.6:1).

Windrow compost produced significantly lower S:H ratios ('plant stress') than vermicompost under similar coir and fertiliser treatments ($p < 0.05$).

S:H ratios were significantly reduced with mixing with coir for both windrow compost and vermicompost ($p < 0.001$, two-way ANOVA), suggesting plant stress were reduced both composts. In the vermicompost derived media this was likely to have been mainly high nutrient levels. In the windrow compost other factors, such as decreased water stress, may have been alleviated, *i.e.*, due to the increased saturated-water-content and reduced air-filled porosity observed after mixing with coir (also observed for vermicompost).

7.4 Conclusions

Both windrow composting and vermicomposting proved to be suitable methods of processing the WPS investigated, resulting large organic matter losses, and final C:N and $\text{NH}_4^+:\text{NO}_3^-$ ratios associated with mature composted products.

Windrow composting resulted in higher rates of organic matter loss, attributed to higher temperatures during processing. Higher losses of nitrogen during windrow composting were attributed to higher processing temperatures and high pH levels, as well as the potential for leachate production.

Vermicomposting resulted in much lower mineral nutrient losses, producing a vermicomposted product with very high levels of available (water extractable) nutrients, in particular, nitrogen and calcium. The availability of nutrients possibly augmented by a lower pH of the vermicompost.

The windrow compost provided a more suitable medium for the cultivation of radish than the vermicompost, derived from an identical WPS; producing rapid germination, plant growth and lower disparities in plant biomass partitioning.

This was mainly attributed to the lower mineral nutrient levels in the windrow compost; a result of higher nutrient losses during the windrow composting process.

The less successful performance of vermicompost was attributed to its very high nutrient contents, although this may be significantly ameliorated by mixing/dilution with coir. Greater dilution of the vermicompost by blending with components low in nutrients, such as coir, may further improve the performance of vermicompost as a plant cultivation medium (*q.v.* Subler *et al.* 1998).

Reduced levels of plant growth in windrow compost mixed with coir indicated the potential of growth limiting levels of certain nutrients, with corresponding nutrient imbalances. However, this was easily alleviated by the addition of a standard commercial fertiliser, possibly suggesting the lower levels of modification are required to transform the windrow compost into a suitable plant growth medium.

Calculating the capacity of each system, windrow composting processed approximately 23 kg WPS $\text{m}^{-2} \text{ week}^{-1}$ (based on the area of windrow alone), whereas vermicomposting processed 4 kg WPS $\text{m}^{-2} \text{ week}^{-1}$ (based on the area of vermicomposting units alone). This equated to a ratio of 5.6:1, comparable to 6:1 estimated by Gellens & Verstraete (1995). However, due to its modular construction (VCUs), the vermicomposting system used in this study, could be stacked into multiple layers, thus processing more material per unit area.

Other practical and economic considerations such as capital, energy and labour inputs required, as well as further quality testing (*e.g.*, plant growth trials using other plant species), and market assessment of the final composted products are required to make a complete evaluation of composting method.

An extra product of the vermicomposting process is earthworm biomass. Further investigation into stocking densities and other parameters that will optimise the economics of vermicomposting, through a balance between waste processing rates and earthworm biomass production rates, are required.

Vermicomposting and windrow composting are clearly very different processes, producing very different composted products with unique qualities. Selection of one system or the other to process WPS will depend largely on the objectives of the process and the requirements of the final product.

8. Main findings and suggestions for further research

8.1 Introduction

This Chapter aims to outline and link the findings presented in preceding Chapters, which is then summarised. Potential areas for further research that have emerged during this study are also suggested.

The economic and environmental implications of each composting process shall be alluded to briefly as little prominence has been given to these issues so far. However, the potential economic and environmental benefits of utilising vermicomposting and windrow composting for the treatment of waste paper sludge was influential to the research as a whole. It is also worthy of note that the research contained within this thesis was jointly funded by The Economic and Social Research Council (ESRC), and a commercial partner (Original Organics) with a vested interest in the findings.

8.2 Main findings

Chapter 1: Introduction

Chapter 1 set out general economic and environmental trends in waste legislation and its implications for organic waste treatment today. A trend towards more integrated waste management was also identified along with the need for more comparative research into waste treatment alternatives.

Previous research into the windrow composting and vermicomposting of organic wastes and their potential economic and environmental benefits was outlined briefly.

The UK paper-production industry was identified as a large source of organic waste (waste paper sludge); and previous research showed waste paper sludge was suitable for both windrow composting and vermicomposting techniques.

A lack of comparative studies between vermicomposting and windrow composting was also identified, and formed the predominant objective of the research.

Chapter 2: Windrow composting of primary and secondary waste paper sludge

Chapter 2 investigated the treatment of waste paper sludges using a pilot-scale mechanically turned windrow-composting system. Two waste paper sludges derived from paper recycling mills were investigated. The waste paper sludges were found to possess very different physicochemical properties that were not immediately amenable to windrow composting. Ways in which the properties of these sludges may be modified with specific amendments to

provide substrates amenable to windrow composting were hypothesised and tested in three large-scale experiments.

Primary waste paper sludge, the product of the primary waste-water treatment, was very low in nitrogen and other mineral nutrients in relation to its carbon content. The addition of these mineral nutrients in an inorganic form (fertiliser) was found to significantly accelerate the stabilisation process, albeit with large losses in nitrogen.

Secondary waste paper sludge, the product of the secondary waste-water treatment, was very high in nitrogen and other mineral nutrients in relation to its carbon content. This sludge was also possessed a high moisture content and lacked the physical structure required for windrow composting. The addition of a bulking agent/carbon source (straw) was found to significantly accelerate stabilisation the composting process.

A third experiment investigated co-composting a mixture of these sludges, which were hypothesised to provide each other with complementary physicochemical characteristics during composting. The co-composting of these two sludges achieved similar rates of stabilisation to those achieved when composting the sludges individually with specific amendments.

Chapter 3: Evaluation of WPS for vermicomposting – earthworm population sustainability and growth

In Chapter 3, the evaluation of vermicomposting for the treatment of the waste paper sludges previously studied (Chapter 2) was not possible due to their unsuitability for earthworm nutrition and survival (Appendix 2).

Because of this, new waste paper sludge was identified with physicochemical properties suitable for earthworm cultivation, and thus vermicomposting. This sludge comprised of an intimate mixture of sludges derived from both primary and secondary waste-water treatment at a cardboard recycling paper mill. This waste paper sludge was also considered suitable for windrow composting, and could therefore be used for a direct comparison between vermicomposting and windrow composting.

Laboratory scale experiments were conducted to assess the nutritional value of this sludge for the cultivation of an epigeic earthworm species (*D. veneta*) commonly used for vermicomposting.

Different stocking densities of earthworms were used to investigate the effect of earthworm population density on the processing (egestion) of WPS, key to its stabilisation by earthworms. The effect of stocking density on the maintenance of an adequate earthworm population for vermicomposting was also investigated.

The increases in earthworm mass at high earthworm densities revealed the high nutritional value of the WPS employed. The highest stocking density of earthworms investigated resulted in the highest rates of WPS processing and organic matter losses, with increased total worm mass and only small levels of earthworm mortality. Earthworm density was found to not only increase the stabilisation of WPS by increased processing (ingestion) but also increased decomposition of non-ingested material.

The knowledge gathered during these experiments regarding appropriate stocking densities for maximum WPS processing, while sustaining an adequate earthworm population, was transferred to a large scale vermicomposting trials (Chapter 5).

Chapter 4: Large-scale composting of a commingled primary and secondary waste paper sludge using an open-air, mechanically turned windrow system

Chapter 4 investigated the use of large-scale windrow composting to process the same WPS identified in Chapter 3 as a suitable substrate for vermicomposting. Due to the high moisture content and compact nature of the WPS, maintaining sufficient aeration of the material was identified as an important parameter when composting this material.

Two methods of aeration were investigated to optimise the windrow composting process: turning frequency and the use of a bulking agent (wood-chips).

It was found that successful composting could not be achieved with frequent turning alone, and that the use of a bulking agent maintained good levels of aeration throughout the composting process. The bulking agent provided aerobic conditions under which significantly more rapid stabilisation and desiccation of the WPS was achieved.

The use of a bulking agent was found to be the most efficient aeration method of those investigated for the mechanically-turned windrow-composting of the WPS under investigation.

Chapter 5: Large-scale vermicomposting of a commingled primary and secondary waste paper sludge using batch vermicomposting systems

Chapter 5 investigated the use of two batch vermicomposting systems with regard to optimising the vermicomposting process when using an identical WPS to that used in Chapter 4.

A multiple batch system investigated the possibility of adding small batches of WPS to a high density of earthworms for periods of 2 weeks (based on the experiences of Chapter 3). The vermicomposted WPS (casts) removed from the vermicomposting units and matured for a further 10 weeks. This system did not fully stabilise the WPS. Casts were only partially stabilised after 2 weeks and lost physical structure during rapid decomposition. This resulted

in the compaction of cast material with a build up of high levels of moisture. This in turn led to the onset of anaerobic conditions and the inhibition of decomposition and nitrification.

A single-batch vermicomposting system, processing a single (larger) quantity of WPS in the presence of an identical density of earthworms for a longer period of 8 weeks, followed by a maturation period of 4 weeks proved optimal. This system resulted in rapid stabilisation of the WPS indicated by rapid organic matter losses and high levels of nitrification. Although, the vermicomposted WPS contained a high moisture content after 8 weeks processing, once it was removed from the vermicomposting units and matured for a further 4 weeks it dried rapidly and stabilised more fully.

The more rapid stabilisation of WPS in the single-batch vermicomposted system emphasised the continued influence earthworms upon organic matter losses after WPS had been processed into casts (ingested). This was consistent with the findings in Chapter 3 that showed the influence of worm activity on WPS other than mere ingestion. This was attributed to the maintenance of aerobic conditions within the processed material by earthworms despite the presence of high moisture levels.

Chapter 6: Windrow-composted and vermicomposted waste paper sludge as components of plant growth media

Windrow composted and vermicomposted WPS obtained from the each optimal windrow composting and vermicomposting system (identified in Chapter 4 and 5, respectively) were used for radish cultivation trials. Composts were mixed with equal volumes of coir or used as the sole component of a plant growth medium. Their performance was assessed in comparison with immature green-waste windrow-compost and a commercially available medium. The performance of plant growth media was assessed in terms of germination rates, plant growth, and the partitioning of plant biomass to shoot at the expense of bulb production (an indicator of radish plant-stress).

The unamended windrow-compost derived media was more suitable for radish cultivation than the unamended vermicompost derived media, which produced similar results to the immature green-waste compost. The windrow-compost was seen to perform as well as the commercial medium, with indications of lower levels of plant-stress.

However, when diluted with coir the windrow compost derived media showed signs of plant nutrient deficiency. When vermicompost was diluted with coir its suitability for radish cultivation improved dramatically, and it was suggested that further dilution would improve the vermicompost derived media further.

Chapter 7: A comparison of mechanically turned windrow composting and vermicomposting in the stabilisation of WPS into a horticultural product.

To make a direct comparison of windrow composting and vermicomposting in terms of the processing (stabilisation) rates and the physicochemical properties of their respective products, results were taken from previous Chapters for further analysis and discussion. To conduct a balanced comparison was possible, results were taken from the windrow composting or vermicomposting system which provided the most optimal conditions for processing WPS in each case.

Both windrow composting and vermicomposting were found to be suitable methods for stabilising the WPS investigated. The WPS was therefore proved suitable for both composting systems and a balanced comparison between these systems was conducted.

Windrow composting was a far more destructive process, resulting in more rapid and greater losses of organic matter and mineral nutrients. Vermicomposting achieved lower rates of organic matter loss, and retained much higher levels of mineral nutrients, especially nitrogen.

Differences between these processes were mainly attributed to physical conditions of each process, such as process temperature. Physicochemical changes effected by these two very different processes were reflected in very different products which was reflected in their performance as plant growth media.

The windrow compost was shown to be a suitable plant growth medium for radish in its unamended form, producing no indications of plant stress or toxicity. This was attributed to its high level of stability and low nutrient content, which was related to high levels of organic matter and nutrient loss during WPS composting. However, this meant it had a low potential as a fertiliser and if diluted by coir required fertilization to produce adequate levels of plant growth.

The vermicompost was not a suitable plant growth medium in its unamended form. This was attributed to its very high nutrient content (especially nitrogen and calcium), related to high levels of nutrient retention during the vermicomposting process. This gave the vermicompost a high potential as a fertiliser. This was shown by higher levels of plant production in vermicompost mixed with coir than coir alone.

8.3 Summary

This study has shown that waste paper sludges can vary dramatically with type of paper manufacture and differing waste-water treatment practices.

This being the case, a number of waste paper sludges were investigated.

Windrow composting was more easily adapted to waste paper sludge processing by the addition of simple structural or nutritional amendments specific to the waste paper sludge under treatment.

Vermicomposting could only process more specifically suited waste paper sludges, limited by the nutritional and environmental conditions required for earthworm survival and growth.

Only once waste paper sludge suitable for both processes was identified, was a 'fair' comparison between processes.

Even when using identical waste paper sludge, equally appropriate for both windrow composting and vermicomposting, these very different composting techniques resulted in different physicochemical changes, reflected in very different composted products possible.

The windrow compost required less modification for its transformation into a successful horticultural product, but the vermicompost showed potential as a horticultural fertilising agent.

It appeared that windrow composting has more potential for the treatment of waste paper sludges produced by paper manufacturing industry. However, due to large variations in the composition of waste paper sludges, in each case the waste must be modified appropriately to optimise its treatment.

8.4 *Further research*

Firstly, due to the large scale of many of the experiments conducted during this study only a narrow range of treatments was possible. It is therefore suggested that further research is required using a broader range of treatments to pinpoint optimal conditions for each composting technique, and transformation of their respective products. For example:

1. Further investigating into different ratios of waste paper sludge to nutritional amendment and bulking agents to maximise WPS stabilisation, and minimise mineral nutrient losses during windrow composting.
2. Investigate different stocking densities of earthworms and different earthworm species to maximise WPS stabilisation and earthworm biomass production during vermicomposting.
3. Explore the transformation of windrow compost and vermicompost into a valuable horticultural product using a wider range of amendments (including coir) at varying levels of dilution. Moreover, to test these products with appropriate ranges of test plant species.

During this study, it was also observed that waste paper sludges contained high levels of aluminium, which may have implications for the use of earthworms produced during vermicomposting, and the potential use of any composted WPS products. It is therefore suggested that:

1. The availability and toxicity of this element should be tested with regard to earthworm and plants, before earthworms or composted WPS be used as commercial products
2. Ways in which the aluminium contents of WPS can be reduced may be investigated, *e.g.* using alternative flocculating agents such as polyelectrolytes.

Finally, a full economic analysis of each composting system is required to make a complete evaluation of each method of WPS treatment. This would include considerations such as capital, energy and labour inputs required, as well as further quality testing and market assessment of the products of each system.

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Appendices

Appendix 1: Origins and composition of waste paper sludges investigated

Primary waste paper sludge (as used in Chapter 2)

This WPS was obtained from an Aylesford Newsprint Ltd recycling paper mill based in Kent, UK. This sludge was produced by the primary waste water treatment of waste water from re-pulping and de-inking processes. These waters were treated with an aluminium sulphate [$\text{Al}_2(\text{SO}_4)_3$] a commonly used flocculating agent (Smethurst 1992) to aid the settling and filtration of suspended solids. The resultant filtered sludge was then screw-pressed to a moisture content of approximately 50% by mass. This WPS was high in inorganic paper fillers, in which spent paper fibres were embedded.

Secondary waste paper sludge (as used in Chapter 2)

This WPS was obtained from an Hook Townsend Ltd. recycling paper mill based in Kent, UK. This sludge was produced by the secondary (biological) waste water treatment of waste waters derived from re-pulping processes. Suspended solids (spent paper fibres and paper filling agents) were flocculated using aluminium sulphate to aid settling and filtration. Firstly, waste waters were treated by activated aeration using the addition of mineral nutrients and oxygen to encourage the microbial reduction of suspended organic matter. The resulting sludge was then pumped to drying beds where moisture content of approximately 80% was obtained. This sludge was also high in mineral inorganic paper fillers as well as high levels of mineral nutrients. The organic matter contained within the pulp consisted mainly of microbial biomass, and little paper fibre remained.

Commingled primary and secondary waste paper sludges (as used in Chapters 3, 4, 5, 6 and 7)

This sludge was obtained from SCA Packaging Ltd., a recycling board and packaging mill. This sludge consisted of an intimate mixture of both primary and secondary waste water treatment processes. Primary waste water treatment consisted of the filtration of spent paper fibres from waste waters derived from board re-pulping processes. To this was added sludge derived from a secondary, activated aeration process similar to that described above. The two sludges were mixed in settling tanks before being passed through a conventional belt-press to obtain a solids content of approximately 20% solids. This sludge possessed a much higher fibre content and lower inorganic matter content due to the fibrous nature of the packaging material recycled. During waste water treatment lower levels of aluminium sulphate flocculant

was used compared to previous sludges treatment processes outlined. This was due to the use of a polyelectrolyte to aid flocculation - Polyaluminium Chloride (PAC).

Appendix 2: Suitability of primary and secondary WPS for vermicomposting

Introduction

The main aim of these experiments was to assess the suitability of three types of waste paper sludge for vermicomposting; primary sludge was obtained from Aylesford Newsprint Ltd., Kent, secondary sludge was obtained from Hook Townsend Ltd., Kent and a commingled sludge was obtained from SCA Packaging Ltd., Kent. The experiments were conducted in order to identify appropriate sludge types for use in later vermicomposting trials, which were to be used as a basis of comparison with windrow composting.

A preliminary laboratory-based experiment initially investigated the sludges for potential worm mortality, while a subsequent medium-scale trial, based on vermicomposting in modular units, was also undertaken. This first medium-scale trial was to evaluate the use of the modular vermicomposting units (VCUs) and to assess the suitability of the primary and secondary sludges for vermicomposting. Subsequent medium-scale vermicomposting trials involved the use of the commingled sludge as feedstock.

Preliminary large-scale windrow composting experiments, using the primary and secondary sludges, were conducted approximately three months before the laboratory-based worm mortality experiments and the medium-scale vermicomposting trials. A particular feature of the vermicomposting assessment procedure was an attempt to determine the sludge characteristics, which were most appropriate for minimising worm mortality and promoting sustainable growth and reproduction. Published research has shown that vermicomposting can be compatible with windrow composting but the nature of the partially composted material fed to worms can have a profound influence on mortality, growth and reproduction (Frederickson *et al* 1997). Hence, a physicochemical evaluation of the fresh sludges and at various stages during windrow composting for the primary and secondary sludges was undertaken.

Physicochemical evaluation of sludges

Edwards (1988) suggests that organic waste feedstock should contain $< 0.5 \text{ mg g}^{-1}$ ammonia, $< 0.5 \%$ salt concentration, and a $\text{pH} < 9$. These figures relate to *E. fetida*, but Edwards (1988) states that these figures do not vary greatly for other species. Lofs-Holmin (1985) in an experiment investigating the toxic effect of polyelectrolyte (non-ionic and cationic) coagulants upon *Allobophora caliginosa*, found mortality occurred after only 3 days in a 50 ppm (highest concentration) cationic polyelectrolyte solution; and 7 days in a 10 ppm cationic polyelectrolyte solution. Kaplan *et al.* (1980) found no toxic effects on worms exposed to 0.1%

concentrations of FeSO_4 , FeCl_3 , $\text{Al}_2(\text{SO}_4)_3$ coagulants, a higher level than that usually measured in sludges.

Table 1 shows the main physicochemical characteristics of the sludges at various stages of composting.

It is apparent that the primary and secondary sludges have the potential to produce quantities of ammonia in excess of 0.5 mg g^{-1} , in fresh form and after 4 weeks composting. Ammonium and pH levels are consistently high for fresh WPS and after 4 weeks 'composting'. At pH 8 approx. 5 % of ammonium will dissociate to $\text{NH}_3(\text{g})$ at 25°C ; pH 8.5 ~20 %; pH 9 ~50 % . Total salts are mostly accounted for by NH_4 contents, although high levels of K are present in secondary WPS. Total salt levels appear to be high enough to be toxic; excluding quantities of salts that were not determined. Electrical conductivity is correspondingly high in secondary WPS. A combination of these factors may produce a higher toxicity than in isolation. A pre-composting period of between 4 and 8 weeks may alleviate toxicity, by which point the addition of worms may become obsolete.

Fresh SCA-WPS possesses relatively low ammonium and pH levels, suggesting little ammonia would be produced. However, ageing SCA-WPS in the laboratory in very thin layers produced strong ammonia odours. Incubator experiments showed worms could be incubated with SCA WPS for a maximum of 2 weeks before showing symptoms of toxicity or stress. Ammonia odours were not detected, but ammonium and other salts may have produced a detrimental osmotic effect. Worms often showed signs of dehydration and, conversely, swelling due moisture retention. The proliferation of a dense white fungal mycelium, at a consistent point during the incubation period (2-4 weeks), appeared to repel the earthworms, absorbing much of the free moisture within the WPS.

After evaluating the effects of composting on the various feedstocks, it was apparent that on advantages would be gained by pre-composting the WPS material, in terms of reducing potential salt toxicity on worms. Fresh WPS material was selected for further vermicomposting experiments.

Table 1: Ammonium, pH, and salt content of WPS feedstocks.

WPS feedstock	Composting Duration	Ammonium content mg g ⁻¹	pH	Total salts (NH ₄ +P+K+Mg) %	Electrical conductivity μS cm ⁻¹
Secondary WPS	0	6.11	8.0	0.73	2588
	4	6.51	8.1	0.82	3038
	8	4.78	8.1	0.61	2013
Secondary + straw	0	5.32	8.7	0.86	2363
	4	1.88	8.3	0.45	1648
	8	0.16	7.7	0.35	1108
Primary + Secondary	0	1.16	8.5	0.14	734
	4	1.07	8.7	0.14	828
	8	0.08	7.5	0.03	546
SCA-WPS + wood chips	0	0.69	7.2	0.40	1350

Mortality trials

Materials and methods

Test units:

These were set up as follows:

0.5 litre (12 cm dia.) containers (n = 5 per treatment)

100 ml bedding; peat + 2% lime (to give pH 6-7); moisture content approx. 75% (Muyima *et al* 1994)

100 g WPS per pot

Stocking density = 8 worms (sub-clitellate) per pot

Incubation temperature 20°C

Treatments:

Primary WPS

Secondary WPS

Primary + Secondary WPS mixture

Commingled WPS

Control (limed peat alone)

Results:

After 14 days incubation, the % survival of worms is given below:

Primary WPS - (95%)

Secondary WPS - (65%)

Primary + Secondary WPS mixture - (81%)

Commingled WPS - (92%)

Control (limed peat alone) - (100%)

The mortality results suggest that the primary, primary + secondary and the commingled sludges would appear to be suitable for larger-scale vermicomposting trials while the secondary sludge should be treated with caution.

Medium-scale vermicomposting trials

The main aim of this experiment was to investigate the effect of earthworms on the decomposition and stabilisation of WPS over 8 weeks active vermicomposting plus 4 weeks maturation.

Materials and methods

The feedstocks selected for the initial trial, were primary WPS, secondary WPS and the mixed primary + secondary mixed WPS. All feedstocks were delivered fresh to the vermicomposting tunnel and used within 24 hours.

Vermicomposting was carried out using individual, modular vermicomposting units (VCUs) as previously described in earlier chapters. Each VCU consisted of a plastic trough (length 0.40 m, width 0.30 m, height 0.25 m) partially filled with 5 litres of moss-peat bedding material, and then covered with 6 kg of beach pebbles (diam. 2–3 cm) to prevent the mixing of WPS and peat during vermicomposting. Active vermicomposting units were stocked with 500g clitellate *Dendrobaena veneta*.

Each WPS treatment comprised 6 active VCUs (containing worms) and 4 control VCUs containing only the WPS feedstock. To each active VCU, 6 kg (~15 litres) of fresh WPS was added; giving a stocking ratio of 12:1 WPS to worms. Inactive VCUs, without worms, (n=4) containing the same quantity of bedding material and fresh WPS were used as controls to monitor the decomposition and physicochemical changes within WPS in the absence of earthworms.

VCU temperatures were maintained at $20.8 \pm 0.7^\circ\text{C}$ inside a thermally insulated polythene-tunnel, using thermostatically controlled under soil cable heating. Temperatures were

monitored using 8×1 m temperature probes connected to a digital data-logger (IceSpy®, © Silvertree Engineering Limited).

Results

It was apparent that after 5 days of vermicomposting, there was 100% mortality within the secondary WPS and the mixed primary + secondary mixed WPS treatments. VCUs containing these treatments contained very high levels of ammonia gas. Worms within the VCUs containing the primary WPS only were showing no signs of stress and no obvious ammonia odour was detected.

The experiment was abandoned after one week. It was assumed that the presence of ammonia gas was responsible for the high worm mortality.

Appendix 3: Vermicomposting units (VCUs)

